Benefits of riparian planting:

A case study of lowland streams in

the Lake Ellesmere catchment

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Freshwater is a globally important resource, yet the continued availability of high quality water is at risk. In New Zealand, around half of lowland water bodies do not meet water quality standards. One of the major threats to water quality in New Zealand is the widespread conversion and intensification of land use.

Since the European settlement of New Zealand, more than 13 million hectares (around 50 % of the total land area) has been cleared and converted to pastoral agriculture. Agriculture is now the dominant land use in most of the middle to lower catchments of New Zealand rivers. This pastoral development has had profound impacts on water quality, aquatic habitats and macroinvertebrate communities. Riparian restoration has been occurring in New Zealand for over 30 years in an effort to minimise the impact on aquatic ecosystems by buffering streams from surrounding land use.

Despite the extent of riparian restoration occurring in New Zealand, little monitoring or evaluation has been undertaken to determine whether planting efforts are achieving their aims. This thesis evaluates the impact of riparian plantings on water quality using a case study on lowland streams in the Lake Ellesmere catchment. A paired catchment design on four river reaches was used to compare restored riparian buffers with control sites upstream. Chemical water quality sampling was used in conjunction with a macroinvertebrate community assessment to provide a comprehensive assessment of water quality.

Riparian restoration was found to have a positive effect on water quality in terms of increasing dissolved oxygen and decreasing turbidity. However, the four plantings that were studied all fail to meet the recommended minimum width of 10 m. This may have limited their effectiveness in protecting water quality, as seen by an increase in conductivity at planted sites, and no changes in other chemical and microbiological factors. Mixed responses were seen in invertebrate community composition, and it is likely that bed substrate, which was unmatched between some paired sites, had a large effect on species present. This research suggests that even narrow planted buffer strips may be effective in improving some water quality variables, and even when no baseline data has been established prior to restoration, monitoring can demonstrate the effectiveness of riparian restoration.

When planning restoration efforts and in the evaluation of their effectiveness, a number of factors need to be considered. This most importantly includes the length and width of buffer strip, time since retirement, stream shade, stream flow and sources of invertebrate colonisers. Ultimately, the effectiveness of riparian planting in protecting water quality requires more planning and a significant monitoring effort in addition to the planting of a stream reach. Finally, it is important to remember that riparian restoration is a long-term task, and the beneficial results it provides will take some time to become apparent. Expectations of landowners and community groups need to be managed, and measurable goals set over a period of years to decades.

Keywords: riparian restoration, stream rehabilitation, monitoring, water quality, freshwater, macroinvertebrates, New Zealand.

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Table of Contents

List of Tables

List of Figures

Chapter 1

Introduction

1.1 Global state of freshwater

Freshwater is a globally important resource. Humans require it for both its direct use for drinking and irrigation, and indirectly for the goods and life supporting services it provides (Postel & Carpenter, 1997). Only a small proportion (2.5 %) of all the water on Earth is freshwater, and only 0.77 % is available for human use in lakes, rivers, aquifers, plants and the atmosphere, with the rest being frozen in either ice caps or glaciers (Postel & Carpenter, 1997).

There are a number of threats to the continued availability of high quality water and the ecosystem services freshwater provides. These threats include the eutrophication and damming of water bodies (Postel & Carpenter, 1997). The results of such threats are seen in New Zealand, where around half of lowland water bodies fail to meet water quality standards (Sustainable Development New Zealand Programme of Action, 2006).

There are also a number of competing demands for freshwater, including farming, hydroelectric power generation, industry, recreation, tourism and cultural uses. It is exceedingly difficult to establish a balance between these often-conflicting uses (Hughey & Taylor, 2008).

Because of the importance of the freshwater resource, there is a need for an international effort to ensure that current needs can be met while the needs of future generations are safeguarded (Ward & Scarfe, 1993).

Globally, coastal freshwater lakes provide important habitat for birdlife and wetland vegetation. These habitats are a threatened ecosystem type, suffering from reduced flushing flows as a result of water extraction and damming and increased nutrient inputs as a result of agricultural development. One such example is Lake Ichkeul in Tunisia (Stevenson & Battarbee, 1991). Riparian restoration is one action that is being taken to address degradation and minimise the effects of these threats.

Since the European settlement of New Zealand, swamps in New Zealand have reduced from about 670,000 hectares to 100,000. These lowland 'wastelands' were all but drained, as they were one of the most productive areas for farming. This decline of more than 85% is one of the most dramatic seen anywhere in the world. Nowadays, the traditional worldview of wetlands as wasteland is being challenged, as ecologists recognise them as one of the highest areas of ecological diversity (Park, 2002).

1.2 Te Waihora/Lake Ellesmere

Te Waihora/Lake Ellesmere is New Zealand's fifth largest lake by area, covering around 20,000 hectares. The Lake Ellesmere catchment drains a total of 256,000 hectares: inland across the Canterbury Plains to the foothills of the Southern Alps, south-west to the Rakaia River, north to the Waimakariri River and the hills of Banks Peninsula (Hughey & Taylor, 2008; Taylor, 1996) (Figure 1.1). Around 40 tributaries, mostly fed by groundwater, drain into the Lake. The main tributaries are the Selwyn, Halswell, L II, and Irwell Rivers, and Harts Creek (Hayward & Ward, 2008; Taylor, 1996). The Lake is brackish, and unusually shallow with an average depth of 1.4 metres and a maximum depth of 2.5 metres (Canterbury Regional Council, 1996; Hughey & O'Donnell, 2008).

The Lake is recognised internationally as a significant site for wildlife and is highly regarded for its conservation value (Hughey & Taylor, 2008; National Water Conservation (Lake Ellesmere) Order, 1990). 167 species of wetland birds have been sighted in its vicinity, and it is a seasonal home for migratory birds. It is recognised nationally as an important site for salt-marsh vegetation, with more than 50 species of these found around the Lake. Te Waihora is also home to 47 species of fish (Hughey & O'Donnell, 2008; Taylor, 1996).

Since the settlement of Canterbury in 1850, the Lake Ellesmere catchment has been gradually and extensively modified (Canterbury Regional Council, 1996; Taylor, 1996). The area surrounding the Lake was drained for pasture, and water races were constructed. Today over 80% of the catchment is in pasture for agriculture. This intensive farming and wetland drainage has had significant impacts on Lake Ellesmere. At the bottom of the catchment, the Lake serves as a trap for nutrients and sediments, which have been increasing with the further intensification of land use (Hughey & Taylor, 2008). The Lake is now considered to be eutrophic, but it does not exhibit traditional characteristics of enrichment. Its shallow nature and frequent strong winds result in a mixing effect where oxygen is transferred through the water column, largely preventing algal blooms (Canterbury Regional Council, 1996; Hayward & Ward, 2008; Hearnshaw & Hughey, 2010; Taylor, 1996). In addition, flows of the contributing rivers to the Lake have dropped significantly due to water extraction.

The Lake is usually separated from the sea by Kaitorete Spit, a large shingle bar. Lake Ellesmere is periodically opened to the sea when the water level is high, by the mechanical cutting of a channel through the shingle bar. This has been done since early European occupation to prevent flooding of farmland. The opening is closed by wave action in southerly storm events (Canterbury Regional Council, 1996; Taylor, 1996).

The Lake is used in a wide range of recreational activities, including fishing, hunting, bird watching, water sports, picnicking, camping, cycling and conservation activities (Booth, 2008). The conflict of values associated with these activities, and between these and the values of agricultural practices have resulted in ongoing debate about the future of the Lake (Hughey & Taylor, 2008).

Te Waihora is also of particular cultural significance to the local Māori tribe, Ngāi Tahu (Taylor, 1996; Te Rūnanga o Ngāi Tahu and the Department of Conservation, 2005). Ngāi Tahu have inhabited the area around the lake for over 40 generations and its natural resources are of fundamental importance to the tribe (Taylor, 1996). This was recognised as part of the Ngāi Tahu Waitangi Tribunal claim settlement in 1996, where the bed of the lake was vested with the tribe (Te Rūnanga o Ngāi Tahu and the Department of Conservation, 2005). Mahinga kai (food resources) harvested around the Lake by Ngāi Tahu include flounder, eels, mullet, whitebait and flax, as well as several bird species (Taylor, 1996). There has been a decline in these species due to changes in habitat and the invasion of exotic competitors (Taylor, 1996). Ngāi Tahu is therefore unable to harvest these resources at the levels they were previously able to, which affects the mana or prestige of the group.

Mauri and kaitiakitanga are also particularly important Māori concepts in relation to water resource management. From the Māori worldview, ensuring the environment and resources are protected is of paramount importance. This protection of the environment is carried out through the exercise of kaitiakitanga, the concept of guardianship, stewardship and protection (Mead, 2003; Royal, 2003). Kaitiakitanga is important in making sure that resources are used in a sustainable way, to ensure their availability into the future.

Mauri refers to the life force, spark of life or essence that is possessed by all living things (Barlow, 1991). All life forms owe their health and continued existence to the mauri that they possess. It is important that the mauri of a resource is not destroyed. Māori are particularly concerned about declines seen in freshwater and the threats that these declines put on the mauri of the resource (Tipa & Teirney, 2003).

Ngāi Tahu are concerned about the declines seen in the Lake in relation to food resources and cultural values, and interested in aiding efforts to restore the Lake. Te Waihora is now managed under a Joint Management Plan between the Department of Conservation and Ngāi Tahu to maintain and enhance its significant values (Hughey & O'Donnell, 2008; Te Rūnanga o Ngāi Tahu and the Department of Conservation, 2005).

It is clear that the problems that have been identified in the Lake Ellesmere catchment with regards to the decline in freshwater quality are not isolated to this area. One of the responses to these declines has been the emergence of non-government organisations concerned with restoring ecosystems and promoting good management practices. An example of one such organisation in the Lake Ellesmere context is the Waihora Ellesmere Trust.

1.3 The Waihora Ellesmere Trust

A co-operative approach has been identified as the most effective way to address the issues surrounding the Lake (Waihora Ellesmere Trust, 2004). The Waihora Ellesmere Trust (WET) was formed as part of this approach in September 2003. WET now has a membership base of more than 100 people including farmers, conservationists, bird enthusiasts, representatives of local Māori groups, recreational users, and local residents. The Trust works with the community and external organisations in order to manage the Lake and its catchment (Waihora Ellesmere Trust, 2004).

In 2003 and 2004 a document titled "A community strategy for the future management of Lake Ellesmere/Te Waihora and its tributaries" was put together setting out the management visions, goals, targets and actions for the Lake (Waihora Ellesmere Trust, 2004). The vision identified in this document is to ensure that Lake Ellesmere is:

> *"A place where healthy and productive water provides for the many users of the lake while supporting the diversity of plants and wildlife that make this place unique.*
>
> *A place of cultural and historical significance that connects us with our past and our future.*
>
> *A place where environmental, customary, commercial, and recreational values are balanced while respecting the health of the resource.*
>
> *A special wide open place for the enjoyment and wonderment of present and future generations.*
>
> *A place of contemplation and tranquility as well as activity, a place just to be."* (Waihora Ellesmere Trust, 2004, p. 5)

The Trust aims to improve the health and biodiversity of the lake and its catchments through promoting better management practices (Waihora Ellesmere Trust, 2009). One action recognised in the community strategy to work towards this goal is to "promote and support riparian plantings" in the catchment (Waihora Ellesmere Trust, 2004, p. 8). In New Zealand, this action is by no means limited to the Lake Ellesmere catchment. Significant rehabilitation projects have been embarked on in other New Zealand catchments including the Motueka and Raglan Harbour tributaries.

Funding and support for restoration programmes through the WET is available from several sources, both regionally and nationally. Environment Canterbury has an Environment Enhancement Fund that can be applied for by groups undertaking restoration projects, and used to assist with the costs of such projects. Environment Canterbury administers the Living Streams programme, which works with local communities to restore rural streams by providing advice and support. The WET has also been allocated funding from the Sustainable Farming Fund and the Community Environment Fund (previously the Sustainable Management Fund). Property owners within the Lake Ellesmere catchment can seek free advice, attend workshops and request financial assistance to enable the planting of native trees from the Waihora Ellesmere Trust (Waihora Ellesmere Trust, 2009).

Although riparian planting has been occurring in the Lake Ellesmere catchment for over ten years, little monitoring or evaluation has been undertaken to determine whether planting efforts are achieving their aims. This also appears to be the case in other stream restoration projects (Bash & Ryan, 2002; Kondolf & Micheli, 1995). Without monitoring, it is not possible for resource managers to demonstrate that the project has achieved its aims (Bash & Ryan, 2002). Project monitoring is also important in providing feedback to improve processes or methods undertaken in the planting and management of the area (Bash & Ryan, 2002).

Monitoring can be defined as the planned and ongoing measurement of environmental factors to identify change over both time and space to achieve clear goals (Goldsmith, 1991). Monitoring is especially important in restoration projects for measuring success. Baseline data makes up an important part of a monitoring programme, enabling comparisons to be made and trends to be established over time (Bash & Ryan, 2002; Spellerberg, 2005). It is also important that monitoring methods are both standardised and repeatable to allow others to follow protocols and ensure trends over time to be detected.

In restoration projects there is often a low requirement to monitor, or monitoring is not undertaken due to a lack of funding, time and experienced persons. Funders have frequently been criticised for the over-direction of funding to the planting aspect of the project. Meanwhile, they place little emphasis on baseline data collection and ongoing monitoring (Bash & Ryan, 2002). It is critical that ways to strengthen the perceived importance of monitoring within projects and increasing funding to allow this should be explored (Bash & Ryan, 2002).

A major indicator of the health of the catchment is the quality of the water. There is a critical need to monitor and evaluate the success of riparian plantings and their role in improving water quality in the Lake Ellesmere tributaries.

1.4 Aims and objectives

This study aims to evaluate the impact of riparian plantings on water quality using a case study on lowland streams in the Lake Ellesmere catchment. This approach is based on an study undertaken by Parkyn et al. (2003) in the Waikato Region, New Zealand. A paired catchment design will be used to compare four restored riparian buffers with unplanted control areas upstream. The four sampling pairs will be located in river reaches on Harts Creek, Boggy Creek and Birdlings Brook (a major tributary to Harts Creek). Water quality will be compared between treatments using temperature, dissolved oxygen, conductivity, nitrogen, phosphorous, turbidity and microbiological counts as well as indices of aquatic invertebrate community diversity. In doing this, baseline water quality and invertebrate community data will be collected and methods for this sampling detailed, allowing further monitoring of the effectiveness of these plantings into the future.

1.5 Thesis outline

This introductory chapter provides background to the thesis. The first section outlines the global state of freshwater, followed by a background to Lake Ellesmere. The Waihora Ellesmere Trust is then introduced, and lastly the aims and objectives of the research are outlined.

The literature review in Chapter 2 begins by presenting the current situation of pastoral agriculture in New Zealand. A discussion about riparian restoration follows, and the emerging trend of using planting to buffer waterways from surrounding land uses is highlighted. Finally, the importance of monitoring restoration efforts and approaches that could be used in such programmes are presented.

The methods by which monitoring was undertaken are outlined in Chapter 3 including field data collection and statistical analyses methods. Site characterisations and assessments of water quality parameters, bacterial counts and macroinvertebrate communities are presented in Chapter 4, followed by a discussion of the main findings in Chapter 5. Lastly, conclusions are presented in Chapter 6.

The definition of restoration as used in this thesis is not absolutely the return of an ecosystem to its pre-disturbance condition (as in Kondolf & Micheli, 1995). The term restoration is problematic, as it is difficult to know when the target should be (pre-Eurpoean or pre-Maori), and often true restoration is not possible due to the presence of new or absence of past species. Instead, a broader definition for restoration as steps taken toward enhancing degraded areas is intended.

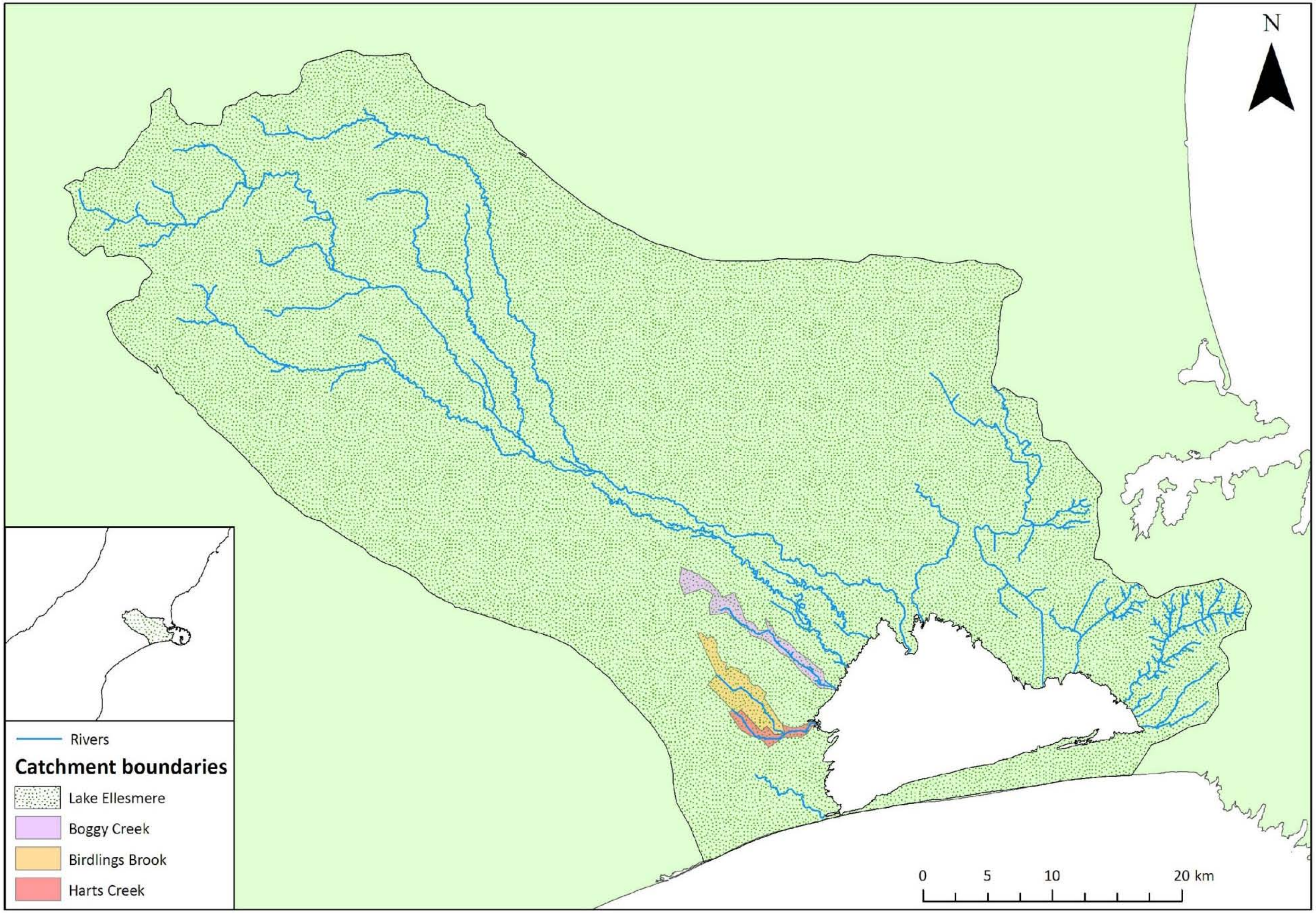

Figure 1.1 The location and extent of the Lake Ellesmere catchment, showing major tributaries and the Harts Creek, Birdlings Brook and Boggy Creek catchments.

Chapter 2

Literature Review

2.1 Introduction

This chapter begins by outlining the widespread land use conversion to pastoral agriculture that has occurred in New Zealand since the arrival of Europeans. The significant impact that this development has had on lowland water quality is identified, and the ability for riparian restoration to buffer aquatic systems from surrounding land use is discussed.

The increasing frequency of riparian restoration worldwide is recognised, however a lack of monitoring such projects worldwide is apparent. The pressure-state-response model is accepted as one way to measure such efforts. Different approaches to monitoring riparian restoration effectiveness, including how to establish a monitoring programme and what to measure are considered. Finally, the chosen methods of monitoring water quality and macroinvertebrate communities discussed in detail.

2.2 Current situation

In New Zealand, over the last 150 years, more than 13 million hectares of land has been cleared and converted to agricultural farming, equating to around 50% of the total land surface (Collier et al., 1995; Quinn, 2000). Pastoral agriculture is now the dominant land use in most of the middle to lower catchments of New Zealand rivers (Quinn, 2000). This conversion has introduced over 60 million grazing animals, mostly sheep and cattle. In addition, fertiliser application totals around 2.2 million tonnes per annum (mainly phosphates) and 1.1 million tonnes of nitrogen per annum is added by nitrogen-fixing plants, such as clover (Quinn, 2000). Furthermore, land use change continues to occur through land drainage, stream channelisation and urbanisation (Collier et al., 1995).

These last 150 years of pastoral development have had profound impacts on water quality, aquatic habitats and macroinvertebrates (Quinn, 2000). The effects of large scale conversion have been well studied and documented in the last 50 years (Storey & Cowley, 1997). In situations where upwards of 30% of a catchment has been converted, a significant change occurs in macroinvertebrate community structure. Specifically, diversity and biomass of clean water taxa are lost and pollution tolerant species increase (Storey & Cowley, 1997). The majority of New Zealand's lowland rivers are now in poor condition due to the land use changes in their catchments (Collier et al., 1995). The Lake Ellesmere catchment is no exception to this decline. The small tributaries are particularly vulnerable to impacts from the adjacent intensive farming, with most streams exceeding guidelines for faecal coliforms and nutrient levels (Hayward & Ward, 2008).

While the agricultural sector is the main cause of this degradation, it has been an essential part of the New Zealand economy (Quinn et al., 1993). In the year ended March 2009 the agriculture and forestry sector provided approximately 12 % of New Zealand's real gross domestic product (GDP) and about 64 % of total exports (Ministry of Agriculture and Forestry, 2009). The agricultural industry is now facing pressure from consumers to use sustainable farming practices to reduce pressures on the environment and live up to New Zealand's "clean and green" image (Collier et al., 1995; Wood & Howard-Williams, 2004).

In 2001, the dairy industry initiated a study in order to address concerns about soil and water degradation and the long term sustainability of the industry (Wilcock et al., 2009). The goal of this study was to establish best management practices to be widely adopted by dairy farmers, resulting in improved environmental performance. The best management practices suggested by this study included: permanent fencing to exclude livestock from waters and planting along water margins to intercept and filter particulate contamination (Wilcock et al., 2009).

Two different sources of pollution have been recognised: point sources and diffuse sources. Point sources of pollution are specifically located, identifiable pipes or drains discharging wastewater and sewage. Diffuse sources of pollution arise from land use activities causing overland flow, where pollutants are picked up and flushed into ground and surface water and leaching, where nutrients and bacteria from travel through to groundwater. While a lot of attention was directed to point sources until the early 1990s, these have now largely been addressed and attention has shifted to diffuse sources, which have proven to be much more difficult to control (Davies-Colley & Wilcock, 2004).

The Resource Management Act 1991 (RMA) is the most important piece of legislation concerning the environment in New Zealand, providing the overall framework for resource management. The RMA encourages sustainable management of New Zealand's resources through ensuring any adverse effects on the environment are avoided, remedied or mitigated (RMA, 1991). Riparian management and planting is one option that resource managers are promoting to minimise the impact of land use change on aquatic ecosystems (Collier et al., 1995).

2.3 Riparian planting

Riparian restoration has been occurring in New Zealand for over 30 years, with the main objective being to buffer the aquatic systems from surrounding land use (Quinn et al., 1993). The riparian zone is identified as the vegetated strip of land extending along the banks of rivers, streams, lakes and wetlands (Parkyn et al., 2000). It is the link between the stream environment and the terrestrial catchment, with a disproportionately large influence on community structure and water quality relative to its proportion of catchment area (Harding et al., 2009; Kauffman & Krueger, 1984; Osborne & Kovacic, 1993). Functions performed by the riparian zone include bank stabilisation, flood control, reductions in peak flow during floods, stream temperature regulation, stock exclusion (reducing bank trampling, defaecation in-stream, stock losses and waterborne illness), filtration of surface runoff reducing in-stream sedimentation and nutrification, provision of organic matter in-stream as a food source, and provision of habitat for fish spawning and adult phases of aquatic invertebrates (Collier et al., 1995; Fennessy & Cronk, 1997; Jorgensen et al., 2000; Kauffman & Krueger, 1984; Lowrance et al., 1984; Osborne & Kovacic, 1993; Parkyn et al., 2003; Parkyn et al., 2000).

Riparian zones planted with native species add to indigenous biodiversity. They can also play an important role as ecological corridors or linkages between other areas of established native planting. These corridors are ecologically important in providing links for dispersal, migration and genetic exchange, nutrient transport and energy flow (Davis & Meurk, 2001). The effectiveness of fenced and planted riparian areas in fulfilling these functions is now widely accepted (Wood & Howard-Williams, 2004).

The close relationship of the riparian zone with the in-stream system makes it a particularly important area for mitigation strategy focus (Quinn et al., 1993). Because of the functions listed above, stream restoration efforts in New Zealand (as well as around the world, including Australia, Japan, Europe and the United States), are focussing on riparian management to buffer the impact of land use on the aquatic environment (Harding et al., 2009; Parkyn et al., 2000; Quinn, 2009). The width, plant composition and plant density are important factors that should be considered when establishing a vegetated buffer (Parkyn et al., 2000).

Indigenous plants initially may not be as vigorous as poplars or willows in stabilising stream banks, however they are better for long term stability and sustainability, because exotic species frequently require ongoing management (Parkyn et al., 2000). Ideally, a planted buffer strip would be self-sustaining and of minimal maintenance; protecting water quality and aquatic habitats, suppressing weed growth and forming a seed bank to allow natural regeneration (Parkyn et al., 2000).

The width of riparian zone that is required to sustain terrestrial and in-stream habitat depends on a number of factors. Between projects, the aims of the planting, channel width, bank slope, vegetation type, position in the stream continuum and hydrological type will vary (Collier et al., 1995; Quinn et al., 2001; Reeves et al., 2004). These variations mean that a 'one size fits all' approach to planting rarely exists, and sites should be considered on a case by case basis (Quinn et al., 2001). Studies comparing multiple widths of planted buffers at the same location have showed that increasing buffer width results in increasing sediment and phosphate removal (Parkyn, 2005). A width of greater than 10 m on either side of the waterway has been recommended as the minimum necessary width for terrestrial biodiversity outcomes and to achieve a self-sustaining strip. Riparian zones with a width of less than 5 m on each side of the waterway are unlikely to support self sustaining vegetation, and weed growth can be a problem (Parkyn et al., 2000). Davis and Meurk (2001) suggest that a buffer between 15 and 20 m wide on either side of the waterway is most likely to support self-sustaining plant populations with minimal maintenance while meeting most aquatic functions.

There is little known about the minimum length of buffer required for stream recovery, however this will greatly depend on the size of the stream and the variable targeted for reduction (Scarsbrook & Halliday, 1998). In spite of the benefits, restoration from the headwaters through to the river mouth is often unrealistic due to the significant cost and private land ownership. Discontinuous restoration is the next best thing, and is likely to mitigate some impacts of land use (Scarsbrook & Halliday, 1998).

Establishing a closed canopy is also recognised as being important, though this will not happen immediately after planting occurs (Parkyn et al., 2000; Wood & Howard-Williams, 2004). Canopy closure is important because it provides shading to the channel, thus moderating water temperature and reducing light levels, minimising water weed establishment and growth (Davis & Meurk, 2001). To help achieve a closed canopy in a reasonably short time frame, and provide ground shading to reduce competition from weed species it is generally recommended that seedlings are planted at a distance of 0.75-1.1 m apart (Parkyn et al., 2000).

In-stream conditions observed at a site are reflective of land use and management practices occurring upstream (Collier et al., 1995; Parkyn & Wilcock, 2004). When attempting to influence a river through planting in the riparian zone, consideration needs to be given to conditions and management practices upstream. Riparian planting should be considered as a secondary restorative measure after controlling the addition of pollutants at their sources (Barling & Moore, 1994).

Timescales are also important to consider in riparian planting (Collier et al., 1995). Riparian management is a long-term task, which requires ongoing maintenance and investment. The beneficial results provided by a riparian zone are not immediate, and may take many years to become apparent. Some studies have indicated that stream conditions may worsen before improvements are seen. This is particularly the case where channel widening occurs following shading (Davies-Colley, 1997). Because of this, there is a need to keep expectations realistic to avoid disappointment (Davies-Colley, 1997). It is also important that realistic targets are set, as it is probably impossible to restore the riparian area and water quality to conditions before land modifications began (Collier et al., 1995).

Two recent studies evaluating riparian restoration effectiveness in Waikato, New Zealand were carried out by Parkyn et al. (2003) and Jowett et al. (2009). The first was carried out on restored areas to evaluate whether the restoration was having an impact on stream health (Parkyn et al., 2003). Nine fenced and planted riparian areas were compared with unbuffered control reaches upstream or in nearby catchments. In general, streams that had planted riparian buffers showed rapid improvements in water clarity and bank stability. Improvements in nutrient loading and bacterial counts were seen at some sites compared with their unplanted equivalents. However changes in macroinvertebrate communities were not seen. Potential reasons for this null finding include a lack of invertebrates to recolonise restored areas, a lack of suitable adult habitat and perhaps that the planted buffers are not achieving habitat restoration goals. However, one stream which had a buffer greater than 50 m wide and more than 25 years old along its entire length did show significant improvements in lowering water temperature and improving invertebrate communities. This may suggest that the timescale required for recolonisation to occur is much longer than expected. Positive changes in macroinvertebrate communities were correlated with reductions in temperature, which suggests that temperature control is important to community restoration (Parkyn et al., 2003).

The second involved monitoring two streams in 1995, 2003 and 2005 prior to and following restoration in 1995/6 (Jowett et al., 2009). The initial restoration involved fencing, building bridges and water troughs and planting a buffer of over 10,000 trees and shrubs approximately 4 m wide on each bank. This study did not carry out water quality monitoring, but suggested that as shade increased, water temperature would decrease and the exclusion of stock would reduce sediment inputs. Over the ten year period, macroinvertebrate communities showed a shift from pollution tolerant to more sensitive species. They suggested that it would take upwards of another 15 years before the planting resembled a forested stream with an overhead canopy, woody debris in-stream providing habitat for aquatic life and native plants outcompeting grasses along banks (Jowett et al., 2009).

2.4 Monitoring riparian restoration efforts

One approach used in the evaluation of environmental problems is the pressure-state-response framework. This approach was created by the Organisation for Economic Cooperation and Development (OECD), and was used by the Ministry for the Environment in the writing of the 1997 State of the Environment report. The pressure-state-response framework evaluates the pressures humans put on the environment, the changes in resource quality and quantity – the state, and the organised responses to address these changes (Ministry for the Environment, 1997).

This framework focuses on the concept of causality, and the idea that the pressures on the environment that humans cause or enhance can be controlled or reduced (Ministry for the Environment, 1997). The pressure-state-response approach is used by Hughey et al*.* (2008)

in their two-yearly survey of people's perceptions of the state of New Zealand's environment.

In the case of riparian restoration, changes in landuse have had significant impacts on water quality, aquatic macroinvertebrate communities and aquatic habitats (Quinn, 2000). One response being used to lessen these effects is riparian restoration. In this thesis, I have accepted that land conversion has significant impacts on waterways, and that riparian restoration is a measure being used to mitigate these impacts. I am interested in assessing the effect that this planting has on the state of water quality.

2.5 Different approaches to monitoring riparian effectiveness

There are a number of ways to establish a monitoring programme for water quality. One could monitor the state of the environment with contrasting riparian treatments, or the state as it changes over time and space. In the study undertaken by Parkyn et al. (2003), restored reaches were compared with unbuffered control reaches upstream on one date. In contrast, Jowett et al. (2009) undertook monitoring of the same area both before restoration was undertaken, and over the ten year period following.

In this context, and within the constraints presented by thesis work, this study focuses on the state of water quality as it responds under different riparian management regimes. While there are a number of limitations with this approach, it is difficult to implement a monitoring programme in any other way given that no baseline data is available for these Lake Ellesmere tributaries prior to riparian restoration occurring. It appears that this has been a common problem in other studies undertaken to establish effectiveness of restoration programmes (Bash & Ryan, 2002).

Having ascertained an appropriate monitoring regime in the context of this thesis to evaluate the current state of the Lake Ellesmere tributary restoration projects, the methods required to assess water quality must be considered. The available options to evaluate water quality include measuring chemical, physical and microbiological water quality, monitoring aquatic macroinvertebrate communities, monitoring fish communities and spawning, measuring cultural health and monitoring people's perceptions.

Traditional methods of measuring stream health are based on chemical, physical and microbiological water quality sampling (Stark & Maxted, 2007). In this study water quality sampling is used in conjunction with a macroinvertebrate community assessment to provide a comprehensive assessment. These parameters are discussed in sections 2.6 – Physical, chemical and microbiological water quality monitoring and 2.7 – Macroinvertebrates as indicators of stream health. Other parameters that could have been measured including the monitoring of fish populations and spawning, Cultural Health Index and State of the Takiwā and monitoring people's perceptions are discussed below.

2.5.1 Fish populations and spawning activity

Fish play an important role at the top of the stream ecosystem. They also provide an important recreational and commercial fishery (UNEP GEMS Water Programme, 2007). In-stream cover from substrate, bank undercutting and overhanging vegetation is important for most fish species (Jowett et al., 2009).

Fish are long-lived and mobile, and at the top of the stream ecosystem they integrate the effects of lower trophic levels. They are also relatively easy to collect and identify to a species level. On account of these characteristics, fish communities and fish spawning activity can be used to show a long-term assessment of water quality (UNEP GEMS Water Programme, 2007).

A 2006 study of trout spawning in the Lake Ellesmere catchment compared recent data with 1980s data in the same catchments. This study showed that there had been an overall decline in trout spawning in the Lake Ellesmere catchment. However, improvements were seen in Harts Creek and Boggy Creek (Taylor & Good, 2006). The brown trout fishery in the Lake has been in decline, from one of the world's best in the 1920s to one of New Zealand's most degraded today (Millichamp, 2008).

In fish monitoring studies, especially where electro-fishing is required, the logistics of training and availability of equipment are difficult; therefore in this study it was decided not to focus on fish populations.

2.5.2 Cultural Health Index and State of the Takiwā

The Cultural Health Index and State of the Takiwā are monitoring tools that link western scientific methods with Māori cultural knowledge (Pauling & Arnold, 2008; Tipa & Teirney, 2003). The Māori Ki Uta Ki Tai (from the mountains to the sea) approach used in resource management is quite different to the European planning paradigm based on a technical scientific approach (Tipa & Teirney, 2003). Iwi need to be able to evaluate waterways in a way that accommodates their values and beliefs, while ensuring reasonable communication and understanding with European resource managers. In this way, the both the Cultural Health Index and State of the Takiwā monitoring tools allow the participation and input of Māori into decision making (Tipa & Teirney, 2003).

The Cultural Health Index is made up of three components: the site status (whether it is traditionally a significant site), mahinga kai values (species present and their use) and cultural stream health (land use, vegetation, sediment, flow, water quality) (Tipa & Teirney, 2003).

Both the Cultural Health Index and State of the Takiwā approaches were applied to Lake Ellesmere in 2007 by representatives of Ngāi Te Ruakikihiki (the local Ngāi Tahu Rūnanga). This assessment showed that the Lake holds significant importance in terms of mahinga kai, despite water quality, land use modification and native vegetation issues. A limitation with the monitoring tool was also identified, whereby values surrounding water and native fish were not directly assessed. The assessors recommended some further refinement of the approach to make it more suitable for assessing lake health (Pauling & Arnold, 2008).

This approach is used by tangata whenua in the evaluation of water quality, and therefore its use was deemed inappropriate in the context of this thesis.

2.5.3 People's perceptions

Local stakeholders' perceptions of water quality in the study area may also be monitored. This involves their views on the change of water quality over time, which may be divided into changes as a result of land use impacts, and changes resulting from riparian restoration. A recent example of public perception being used to evaluate water resources was undertaken by Kerr and Swaffield (2007). This study evaluated the amenity values of spring fed rivers by exploring the ways in which key stakeholders perceive the tradeoffs of water allocation.

This thesis has a focus on the physical aspects of planting. A perception study would not directly measure the impacts of riparian restoration on their target, the waterways. In addition, surveys of public opinion are complex and lengthy and therefore, it was decided not to focus on perception monitoring in this study.

2.6 Physical, chemical and microbiological water quality monitoring

A major part of the overall health of the catchment is the quality of its water. Important variables in the chemical, physical and microbiological measurement of water quality include pH, temperature, dissolved oxygen, conductivity, nitrogen, phosphorous, turbidity and bacterial counts. There is general agreement on the use of these characteristics to determine water quality.

Different accepted standards and guidelines have been set for a number of these variables by different authorities at national and local level. Examples of these authorities include the Australia and New Zealand Environment and Conservation Council (ANZECC) (ANZECC, 2000), the Ministry for the Environment (as in Davies-Colley & Wilcock, 2004), Regional Councils for example the West Coast Regional Council (as in James, 1999) and in the Third Schedule of the RMA (1991).

It is important to note that the ANZECC guidelines are trigger values for the protection of aquatic ecosystems and recreational values. Even when trigger values are exceeded, it does not mean that recreation can no longer occur, or that the ecosystem is being damaged, but is merely an indication that a problem may be occurring (Ministry for the Environment, 2007). The stringent nature of guidelines is necessary as these are designed to provide an early warning to ecosystem stress, although exceeding the trigger values is not conclusive of an adverse outcome (Milne & Perrie, 2006). While the ANZECC guidelines are presented for different parameters below, these ranges are often considered to be very strict, and even healthy waterways often exceed their trigger values (Milne & Perrie, 2006).

The Third Schedule of the RMA (1991) (attached as Appendix 1) sets minimum standards for 11 classes of water. If a Regional Council classifies a body of water as being one of the specified classes, the requirements in the Third Schedule are the minimum level required unless a more stringent standard is put in place (Christensen & Jones, 2011).

In addition to these, water quality scientists recommend that quality guidelines are best set by local data. Catchment specific accepted levels should be derived from at least two years of regular (monthly) sampling (Milne & Perrie, 2006), but no such guidelines could be found for the Canterbury Region.

2.6.1 pH

Testing of pH is one of the most common and important tests in water chemistry (Eaton et al., 2005). The pH measures the acidity or alkalinity of a water sample, based on the concentration of hydrogen (H+) and hydroxide (OH-) ions (Hoare & Rowe, 1992). It is measured on a scale of 1 – 14, where 7 is said to be neutral. When there are more H+ than OH- ions in solution, the water has a pH below 7, and is said to be acidic. When there are more OH- than H+ ions, the water has a pH above 7 and is alkaline (Eaton et al., 2005). pH is measured on a logarithmic scale, meaning that the change between two units is ten times the change in one. Because of this, the further the pH from neutral, the greater the impacts of any decrease of an acidic substance or increase of an alkaline (Kotz et al., 2006).

Stream life is adapted to a certain range of pH levels, and water pH must stay within that range to support aquatic life (Waterwatch Victoria, 1996). Changes in pH affect the solubility and speciation of some compounds, which impacts on a compounds bioavailability and toxicity (Davies-Colley & Wilcock, 2004). pH changes also affect enzyme function and membrane processes in cells (Kotz et al., 2006).

The pH of a stream is naturally affected by the geology and soils of the catchment, salinity, photosynthetic and respiration rates of plants and algae and rainfall. The pH is also affected by human activities such as land practices in agricultural areas, discharges of waste and air pollution (Waterwatch Victoria, 2009). Water is said to be of excellent quality when pH ranges between 6.0 and 8.5 (James, 1999). The ANZECC trigger values for pH in lowland rivers are 7.2 (lower limit) and 7.8 (upper limit) (ANZECC, 2000).

2.6.2 Water temperature

Temperature controls the rate that chemical, physical and biochemical processes can occur (Davies-Colley & Wilcock, 2004). In situations where temperatures are very low, these processes slow down. As temperature increases, process rates speed up until they reach very high temperatures, where bacteria, plants and animals may die (Davies-Colley & Wilcock, 2004). Water temperature also affects dissolved oxygen levels, where cooler water can dissolve more oxygen than warmer water (Waterwatch Victoria, 1996).

Temperature has a significant impact on aquatic invertebrate and fish life, as each species tolerates a certain range of temperatures (Cox & Rutherford, 2000). As temperatures increase within this range, the resilience of organisms to stressors is lost (Davies-Colley & Wilcock, 2004). Outside their optimum temperature range, organisms are likely to be outcompeted by a species that can better tolerate the new range (Waterwatch Victoria, 2009). In addition, sensitive species are often unable to survive in areas where the water temperature exceeds around 20 ^{o}C.

Water temperature tends to fluctuate daily around a seasonal mean, driven by solar radiation (Davies-Colley & Wilcock, 2004). Consequently, temperature values vary with the time of the day the sample is taken. Sources of heat in water include groundwater and point source inflows, sunlight, absorption from air, and bacterial breakdown. Water temperature is usually lowered by the presence of overhanging riparian vegetation shading the channel (Environment Canterbury, 2009).

There is no specified guideline for water temperature, but it is generally accepted that temperatures above 20 ^{o}C will have detrimental impacts on aquatic life (Quinn & Hickey, 1990).

2.6.3 Conductivity

Conductivity measures the ability of a solution to carry an electric current (the presence and concentrations of ions in solution), in micro siemens per centimeter (µS/cm) (Eaton et al., 2005). Conductivity levels are affected by catchment geology and soils, land use activities, as well as flow variations (Waterwatch Victoria, 1996).

Some level of dissolved salts is necessary for the growth of aquatic organisms, however excessive levels may be toxic (Waterwatch Victoria, 2009). To ensure the protection of aquatic ecosystems, the Australian recommended upper limit for conductivity is 1,500 µS/cm (Waterwatch Victoria, 1996). Given that no published upper limit for conductivity could be found for New Zealand, the Australian guideline was considered most reasonable alternative standard for comparison. However, it should be noted that New Zealand does not have such severe salinisation issues as Australia, and it would be very unlikely for a New Zealand freshwater bodies to reach this limit.

2.6.4 Turbidity

Water turbidity is a measure of the scattering of light affected by the presence of suspended matter within a sample (Eaton et al., 2005). This suspended matter consists of silts and clays (inorganic), and detritus (organic) (Kotz et al., 2006). High turbidity is indicated by a murky water sample, and low turbidity indicates a clean sample (Environment Canterbury, 2009). Turbidity is measured in nephelometric turbidity units (NTUs), which measure light transmission and scattering in a sample (Davies-Colley & Wilcock, 2004). Turbidity is commonly used as a rough index of the fine suspended sediment content of the water (Davies-Colley & Smith, 2001).

As the catchment gradient declines and water movement slows, some of the particles that are kept in suspension will drop (Waterwatch Victoria, 1996). Turbidity in rivers comes from catchment and stream bank erosion. Human activities such as agricultural practices, forestry and urbanization all increase sediment loading in water bodies (Waterwatch Victoria, 2009). This causes gravelly stream bottoms to be silted up, pools to be filled with fine sediments and light penetration to be reduced (Kotz et al., 2006). High turbidities affect aquatic life, as it makes it more difficult for invertebrates, birds and fish to locate food, it interferes with oxygen uptake by clogging fish gills and as it settles it reduces the available stream bed habitat for fish and invertebrate breeding (McDowell & Wilcock, 2008). Turbidity also has impacts on fishing, as low turbidity is important for aesthetic and safety aspects of recreational use (Main & Lavendar, 2003). High turbidities often indicate the presence of other pollutants, notably dissolved phosphorus and the bacterium *Escherichia coli* (Davies-Colley & Smith, 2001).

Water is deemed to be of excellent quality when turbidity ranges between 0 – 2 NTUs (Davies-Colley & Wilcock, 2004). The ANZECC upper limit trigger value for lowland rivers is 5.6 NTUs (ANZECC, 2000).

2.6.5 Dissolved oxygen

Dissolved oxygen is a measure of the amount of oxygen dissolved in water, measured in milligrams per litre, or as percentage saturation (Eaton et al., 2005). Dissolved oxygen is important for invertebrate and fish respiration (Davies-Colley & Wilcock, 2004). As dissolved oxygen levels decrease, sensitive fish and invertebrate species are lost.

Dissolved oxygen primarily comes from interactions with the atmosphere on the surface of the water body, but is also increased by plants photosynthesizing (Hoare & Rowe, 1992). Levels are affected by how fast oxygen can enter the water, and the rate that it is used

(Waterwatch Victoria, 1996). Oxygen enters the water more rapidly in shallow, fast moving rivers when compared with deeper, slower flowing rivers (Waterwatch Victoria, 2009). Oxygen-consuming life such as fish, macroinvertebrates and bacteria tend to deplete the dissolved oxygen of the water, so if re-aeration does not occur, levels can drop rapidly (Davies-Colley & Wilcock, 2004).

Dissolved oxygen is also affected by the time of day that sampling is undertaken, as it may vary widely over the course of 24 hours, even in pristine rivers. Diurnal fluctuations occur because during the day photosynthesizing aquatic plants produce oxygen, so the amount increases (Hoare & Rowe, 1992). During the night this photosynthesis does not occur due to the lack of sunlight, though organisms are still using up oxygen in respiration (Main & Lavendar, 2003). In eutrophic conditions these fluctuations are exaggerated, as there is more photosynthesis during the day (often causing concentrations to reach supersaturation), and more respiration during the night.

A dissolved oxygen saturation of 80 % or above is recommended to maintain aquatic life (Davies-Colley & Wilcock, 2004). This level is specified in the RMA as the minimum standard for required for water quality classes aquatic ecosystems (AE), fisheries (F), fish spawning (FS) and gathering or cultivation of shellfish for human consumption (SG) (Appendix 1). The ANZECC trigger values for lowland rivers are 98 % saturation (lower limit) and 105 % saturation (upper limit) (ANZECC, 2000).

2.6.6 Nutrient enrichment

Nutrients such as nitrogen and phosphorous occur naturally in water and are essential for all life (Hoare & Rowe, 1992). Phosphorous is important in the energy transfer in the cells of plants and animals (ADP-ATP process), and is a central component of lipids in cell membranes, DNA and RNA (McDowell & Wilcock, 2008; Monbet & McKelvie, 2007). Nitrogen is the central component of proteins and can form in excess of 10 % of an organisms dry weight (McDowell & Wilcock, 2008; Monbet & McKelvie, 2007). Their in-stream concentration is measured in milligrams per litre (mg/L) (Hoare & Rowe, 1992).

Either nutrient can act as a limiting factor when the other nutrient concentrations are high, but when both are over-abundant this can cause excessive plant growth, algal blooms and eutrophication (Davies-Colley & Wilcock, 2004). Dissolved inorganic forms of nitrogen and phosphorous are particularly concerning for in-stream habitats, as they are immediately available for use by aquatic plants (Hoare & Rowe, 1992).

The forms of phosphate found in waterways tend to bind to clay minerals in suspension, whereas nitrates remain soluble. Because of this, human nitrogen inputs are much more likely to reach water bodies and are easily transported through them. This results in nitrate levels being almost always higher than phosphates (Waterwatch Victoria, 2009). Because of this, phosphorus is often the limiting factor in waterways, it limits growth at over three-quarters of the 900 sites in New Zealand regularly sampled by Regional Councils (McDowell & Wilcock, 2008). Elevated phosphate concentrations are of particular concern and can result in algal blooms and nuisance plant growth (Monbet & McKelvie, 2007). Nutrient limitation occurs when the ratio of nitrogen:phosphorus differs from 7:1, where if more than seven nitrogen units are present per phosphorus unit, phosphorus is the limiting factor (McDowell & Wilcock, 2008). This ratio must always be considered in the context of the overall concentrations of both nutrients. An example of this is if concentrations of both nutrients are high, then eutrophication can occur even if one is limited (McDowell & Wilcock, 2008).

Nitrogen and phosphorus naturally enter waterways through rock weathering and decomposition processes. These compounds are also found in fertilizers, effluent, cleaning compounds, soil sediments and plants. They may enter waterways from runoff, stock access and point discharges (Environment Canterbury, 2009). Compositions of both nutrients can be expected to be much higher in wet weather conditions when rapid surface runoff is occurring (Waterwatch Victoria, 2009).

The Ministry for the Environment has set guidelines for dissolved inorganic nitrogen levels to be between 0.04-0.1 mg/L and dissolved reactive phosphorus to be between 0.015-0.03 mg/L (Davies-Colley & Wilcock, 2004). The ANZECC trigger values for lowland rivers are 0.444 mg/L for nitrate-nitrogen (NO_3-N) and 0.01 mg/L for dissolved reactive phosphate (upper limits) (ANZECC, 2000).

2.6.7 Microbiological counts

A diverse range of bacteria occupy freshwater habitats (Chigbu & Sobolev, 2007). While it is impossible to measure every possible disease causing bacteria that could be in waterways, there are several species that are used as indicators of faecal pollution and the possibility of the presence of faecal pathogens. Microbiological counts are used to determine the presence and counts of these indicator species in the water to assess health risks (Chigbu & Sobolev, 2007).

Faecal coliforms and their main constituent species *E. coli* occur naturally in the gut of warm-blooded animals, and their presence shows the potential occurrence of faecal and other pathogenic material (Chigbu & Sobolev, 2007). There is no simple way to tell if *E. coli* found in a river is from animal or human faeces, but where animal faeces are present other bacteria including *Campylobacter* are likely (Davies-Colley & Wilcock, 2004). Due to this, *E. coli* is now commonly used in monitoring programmes as an indicator of faecal contaminants (McDowell & Wilcock, 2008). Salmonella is found in both warm- and cold-blooded animals, and is usually associated with bird life. Salmonella poses a health risk to humans, where it can cause food poisoning if ingested.

There are a variety of ways bacteria can enter the water: through sewage overflows, poorly treated sewage, septic tanks, stock access to unfenced waterbodies, stock crossings, land spreading of effluent, storm water, runoff from agricultural land, wallows and feed-pads and from wildlife living in or around water bodies (McDowell & Wilcock, 2008). Following periods of rainfall and in floods the bacterial count may be greatly increased due to increased mobilisation of sediment by overland flows (McDowell & Wilcock, 2008).

Bacterial counts are measured in colony forming units per 100 mLs (CFUs/100 mL). In the membrane filter method, each dot on a bacterial plate is a CFU, representing thousands of bacteria. Levels of various bacteria in water are used to determine whether the water is fit for drinking, shellfish and fish collection and for contact recreational uses. For contact recreation less than 150 CFUs/100 mL of coliforms and less than 126 CFUs/100 mL of *E. coli* are recommended by the ANZECC (Davies-Colley & Wilcock, 2004). The Ministry for the Environment and Ministry of Health (2003) also have set guidelines using *E. coli* to assess the risk of faecal contamination in freshwater recreational areas. When the count is less than 260 CFUs/100 mL this is deemed an acceptable level, from 260 – 550 CFUs/100 mL the site should undergo an assessment to identify the source, and above 550 CFUs/100 mL in addition to a survey signs should be erected and media used to warn the public (Ministry for the Environment & Ministry of Health, 2003).

2.6.8 Summary of accepted levels of parameters used to measure water quality

Table 2.1: Summary of accepted levels of physical, chemical and microbiological parameters used to measure water quality

Variable	Acceptable level	Source
pH	6.0 – 8.5	James (1999
	7.2 – 7.8	ANZECC (2000)
Temperature	< 20 °C	Quinn & Hickey (1990)
Conductivity	< 1,500 uS/cm *	Waterwatch Victoria (1996)
Turbidity	0 – 2 NTUs indicates excellent quality	Davies-Colley & Wilcock (2004)
	< 5.6 NTUs	ANZECC (2000)
Dissolved oxygen	> 80 % saturation	Davies-Colley & Wilcock (2004)
	98 ≤ x ≥ 105 % saturation	ANZECC (2000)
Nitrogen	0.04 – 0.1 mg/L	Davies-Colley & Wilcock (2004)
	< 0.444 mg/L	ANZECC (2000)
Phosphorus	0.015 – 0.03 mg/L	Davies-Colley & Wilcock (2004)
	< 0.01 mg/L	ANZECC (2000)
Coliforms	150 CFUs/100 mL	Davies-Colley & Wilcock (2004)
E. coli	126 CFUs/100 mL	Davies-Colley & Wilcock (2004)

* Note: as New Zealand does not have such severe salinisation issues as Australia, and it would be very unlikely for a stream to reach this conductivity limit.

2.7 Macroinvertebrates as indicators of stream health

Aquatic macroinvertebrates are small animals that spend most of their lifecycles in rivers or lakes. A number of different macroinvertebrate taxa are found in New Zealand streams, including insects, crustaceans, worms, flatworms and snails (James, 1999).

Aquatic invertebrates fulfill an important role as primary producers in rivers. They eat algae and convert it to energy for other species, (Stark et al., 2001) providing an important link in river food webs (Winterbourn, 2004).

Macroinvertebrates are also commonly used as an integrated assessment of water quality, indicating ecological condition at the site (Bain et al., 2000; Boothroyd & Stark, 2000). This is possible because aquatic invertebrates are a diverse group of species, which have varying long term tolerances to conditions and respond predictably to habitat disturbances and pollution (Boothroyd & Stark, 2000; James, 1999; Winterbourn, 2004). Mayflies (Ephemeroptera), caddisflies (Trichoptera) and stoneflies (Plecoptera) are generally considered to be sensitive to pollution and habitat disturbance, whereas chironomids (Chironomidae), molluscs (Mollusca) and crustaceans (Crustacea) are more tolerant (Hickey & Clements, 1998).

Traditional methods of measuring stream health are based on chemical water quality sampling. One drawback of using this method is that the result only reflects the conditions at the point in time when the sample is taken (Stark & Maxted, 2007). Macroinvertebrates have a life cycle of at least a year, and are confined to the stream area being sampled. They are also normally abundant in stream systems and easily sampled and identified (Boothroyd & Stark, 2000; Stark et al., 2001).

There are some problems with using aquatic invertebrates as indicators. It is difficult to determine the cause of community change when several factors may be having an impact (Boothroyd & Stark, 2000). There is high spatial variation in aquatic communities so areas need to be sampled rigorously to ensure valid data are collected (Boothroyd & Stark, 2000).

Monitoring surveys usually result in a large amount of complex data being collected. It is useful to be able to summarize this information to present it clearly and concisely (Stark, 1984). A number of easy to understand diversity, community similarity and biotic indices have been developed to interpret monitoring data (Boothroyd & Stark, 2000).

Diversity indices are a mathematical expression of species richness, evenness in distribution of individuals among taxa, and invertebrate abundance. They describe community responses to a particular environment. These include Shannon-Weiner's index, Margalef's index and Simpson's index. Comparative indices compare two or more populations to identify spatial differences between them (Boothroyd & Stark, 2000). They include Jaccard's index, Sorensen's index and Pinkham-Pearson index.

Biotic indices incorporate a pollution tolerance score to diversity indices. Invertebrate tolerance scores have been assigned to taxa based on data showing their ability to cope in different ranges of water quality (Boothroyd & Stark, 2000). These include the macroinvertebrate community index (MCI and its quantitative and semi-quantitative versions) and the stream health monitoring and assessment kit (SHMAK).

Community taxonomic composition can also be used to indicate ecosystem health (Bain et al., 2000). The EPT taxa index is the percentage of a sample made up of taxa in the insect groups Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies). These three insect orders are considered to be clean water taxa, inhabiting water only of an excellent quality. The greater the diversity of the EPT taxa, the better the water quality and habitat at the site (Bain et al., 2000).

2.7.1 The Macroinvertebrate community index (MCI)

The Macroinvertebrate community index (MCI) was proposed by Stark (1985) to assess organic enrichment in stony riffles in New Zealand streams. The concept for the MCI was derived from a macroinvertebrate-based score system developed by the British National Water Council's Biological Monitoring Working Party (BMWP) (Stark, 1985). The MCI requires presence-absence data, and is mainly based at a genera level (Boothroyd & Stark, 2000). It also has quantitative (QMCI) and semi-quantitative (SQMCI) equivalents, which require all animals to be counted or an estimate of abundance respectively (Boothroyd & Stark, 2000).

In the MCI, each taxon is given a score from one to ten based on their pollution tolerances. A score of one indicates that the taxon is very tolerant of pollution, and a score of ten indicates that the taxon is very sensitive to pollution. MCI scores for New Zealand taxa can be found in Appendix 2. A site score is calculated by adding the individual taxa score, then the MCI value can be calculated by dividing the site score by the number of scoring taxa, and then multiplying by 20 (Figure 2.1).

$$\mathrm{MCI} = \frac{\sum_{i=1}^{i=S} a_i}{S} \times 20$$

Figure 2.1: MCI calculation (Stark, 1985).

Overall MCI scores range from 0 – 200. A score between 0 – 80 indicates probable severe pollution, 80 – 99 indicates probable moderate pollution, 100 – 119 indicates moderate enrichment and 120 – 200 indicates clean water (Stark & Maxted, 2007).

The main objectives of the MCI are to reduce the time and expense associated with sampling macroinvertebrates, and to make data collected easily understandable and comparable

(Stark & Maxted, 2007). However, sometimes quantitative data are required and in this situation the Quantitative Macroinvertebrate Community Index (QMCI) or the Semi-Quantitative Macroinvertebrate Community Index (SQMCI) may be required.

The QMCI and SQMCI both incorporate the abundances of invertebrates found, the QMCI as the number observed and the SQMCI on a scale of coded abundances from rare to very very abundant (Stark & Maxted, 2007). These two indices respond to changes in both the taxonomic and numerical composition of communities, whereas the MCI only responds to changes in taxonomic composition (Stark & Maxted, 2007).

2.7.2 Use of the Macroinvertebrate community index in New Zealand

Use of the MCI in New Zealand is well established, having been used in a range of studies undertaken from the early 1990s.

Quinn and Hickey (1990) felt that the MCI was more useful as an indicator of water quality than species diversity, richness or percentage EPT taxa. Studies have found that the MCI/QMCI scores were reduced significantly in areas that were in pasture compared with native or pine forest, or in catchments where the channel had been straightened (Collier, 1995; Quinn et al., 1997; Quinn et al., 1992; Scott et al., 1994). Collier et al. (1998) found positive correlations between the MCI and stream shade and the MCI and percentage of catchment in native forest. Stark (1993) found that hand-kick net sampling was more precise for estimating MCI values than Surber sampling. Stark (1993) also recommended that the MCI be used over the QMCI where cost effectiveness is a priority, as it more reliably estimated from fewer samples.

2.7.3 Advantages and disadvantages of using the Macroinvertebrate community index

The MCI shows changes in the community’s taxonomic composition but not numeric composition, potentially making it less sensitive to changes in composition than the QMCI and SQMCI (Boothroyd & Stark, 2000). However, it has been suggested that the MCI might be a more sensitive index of water enrichment (Quinn & Hickey, 1990).

The MCI, SQMCI and QMCI have gained some criticism when applied to pollution types other than organic/nutrient enrichment as they do not detect heavy metal pollutants (Hickey & Clements, 1998). They have also attracted criticism when applied to silty or macrophyte covered riverbeds, where they are less able to detect changes in habitat quality. The invertebrates that typically inhabit soft substrates have low MCI scores. Therefore it is extremely unlikely that a stream with a soft bed would have a MCI greater than 130, even if water quality is excellent (Boothroyd & Stark, 2000).

Interpretation of the MCI and its variants is not always straightforward. When invertebrate data are collected on their own, with no information on water quality, it is difficult to determine the extent and causes of the pollution. On account of this, it is advisable that invertebrate samples be collected in conjunction with water quality information. Water quality and invertebrate sampling are complementary in nature, where sampling of water quality shows condition at that point in time and invertebrate sampling shows condition over a longer time period. There are also a number of factors that can influence the final index value, including water quality, sediment, flooding/low flow frequency and shading (Boothroyd & Stark, 2000).

2.8 Other important factors affecting stream health

2.8.1 Stream shade

Solar energy drives primary production in stream, as well as providing illumination and increasing temperatures (Davies-Colley & Payne, 1998). In situations where warmer temperatures are combined with available nutrients, prolific plant and algae growth can occur. Stream shade can be increased by riparian planting, which then regulates temperature and stream plant growth (Reeves et al., 2004). Small streams are more at risk of heating due to their shallow depth (Reeves et al., 2004). Therefore, riparian shade is most effective at temperature regulation in small streams, where a closed canopy can be achieved (Reeves et al., 2004).

2.8.2 Aquatic macrophytes

Aquatic macrophytes are large submerged, floating and emergent plants that grow in rivers and lakes (Eaton et al., 2005). Common examples of aquatic macrophytes found in New Zealand streams include *Elodea canadensis* (Canadian pond weed), *Lemna minor* (duckweed), *Mynophyllum propinguum* (water milfoil), *Rorippa nasturtium* (watercress), *Mimulus guttatus* (monkey musk) and *Ranunculus tricophulllus* (water buttercup). Growth of aquatic macrophytes is promoted by nutrients and sunlight. Dense stands of macrophytes can have undesirable impacts on water quality, including: causing oxygen and pH to fluctuate, impeding currents, encroaching on channels and inducing sedimentation.

However, in some situations aquatic macrophytes can also be beneficial. Some macrophytes are particularly effective at taking up nutrients from the sediment and water, controlling nutrient concentrations and providing habitat and food for aquatic macroinvertebrates.

2.9 Summary

This literature review begins by identifying the large-scale land use conversion that has occurred in New Zealand, and the impact that this has had on water quality. The increasing trend in riparian restoration efforts to protect streams from their surrounding land uses is discussed. Despite the increase in riparian restoration projects, a lack of monitoring such projects worldwide is apparent. Different approaches to monitoring riparian restoration effectiveness, including how to establish a monitoring programme and what factors to measure are considered. Finally, the most suitable methods to measure riparian restoration effectiveness in this situation: chemical water quality and macroinvertebrate communities were discussed in detail.

Chapter 3

Methodology

3.1 Introduction

This chapter outlines the methodologies undertaken in this study to evaluate the impact of riparian plantings on water quality. Firstly, it outlines the study design used, followed by the field data collection techniques including site characterisation protocols, water quality and aquatic macroinvertebrate sampling methods. Limitations of the chosen methods are then outlined. Lastly the statistical analyses methods are detailed.

3.2 Study design

There are a number of ways a monitoring programme to evaluate riparian restoration effectiveness could be established, as discussed in section 2.5 - Different approaches to monitoring riparian effectiveness. This study is based on an earlier study of the role of restored riparian areas in improving stream health carried out by Parkyn et al*.* (2003) in the Waikato region. This approach was chosen given the similarities that no baseline data had been collected prior to riparian restoration. The aim of the Parkyn et al*.* (2003) study was to determine whether riparian restoration was achieving improvements in stream health, including water quality and aquatic invertebrate populations. In addition, this thesis also incorporates multiple sampling dates to ensure data consistency.

Four reaches where riparian buffers have been planted were used to evaluate the effectiveness of riparian planting on water quality in the Lake Ellesmere catchment. The reaches are located on Boggy Creek, Birdlings Brook and at two locations on Harts Creek. Sites were selected following discussion with the Waihora Ellesmere Trust and the Harts Creek Streamcare Group. The criteria for selection were based on the buffered areas being best examples of riparian restoration in the area. While the plantings failed to meet 10m minimum recommended width guidelines and were of a wide age range, they were praised by landowners, contractors and local government officials as exemplars of restoration. The locations of the sampling sites were limited by the willingness of landowners to grant access to their properties. As a result, while control and buffer reaches were on the same stream, in some pairs control sites were not located directly upstream of the buffered reach.

The four restored sites are Boggy Creek at Volckman Road (Figure 3.1, site 1 Figure 3.5), Harts Creek downstream of The Lake Road (Figure 3.2, site 3 Figure 3.5), Harts Creek downstream of Lochheads Road (Figure 3.3, site 5 Figure 3.5) and Birdlings Brook at Beethams Road (Figure 3.4, site 7 Figure 3.5). Each of these sites have buffer zones that have been planted with native vegetation (Table 3.1). Three of the buffer zones are fenced to exclude stock, and the fourth site is not grazed (Harts Creek at Lochheads Road).

Figure 3.1: Planted buffer on Boggy Creek (paired reach 1).

Figure 3.2: Planted buffer on Harts Creek downstream of The Lake Road (paired reach 2).

Figure 3.3: Planted buffer on Harts Creek downstream of Lochheads Road (paired reach 3). Photo courtesy of Hamish Rennie.

Figure 3.4: Planted buffer on Birdlings Brook (paired reach 4).

Figure 3.5: Sampling sites.

Sites 1 & 2: Boggy Creek (paired reach 1), Sites 3 & 4: Harts Creek The Lake Road (paired reach 2), Sites 5 & 6: Harts Creek Lochheads Road (paired reach 3), Sites 7 & 8: Birdlings Brook (paired reach 4).

See Appendix 3 for grid references and Appendix 4 for aerial photos of site locations.

Table 3.1: Buffer reach characteristics.

	Buffer age (years since planting)	Buffer length (m above study site)	Buffer width (m)	Vegetation composition
Boggy Creek	2	2,200	5	*Carex secta* (sedge), *Cordyline australis* (cabbage tree), *Cortaderia toetoe* (toetoe), *Sophora microphylla* (kowhai), *Griselinia littoralis* (kapuka), *Kunzea ericoides* (kanuka), *Plagianthus regius* (lowland ribbonwood), *Pittosporum tenuifolium* (pittosporum), *Pittosporum eugenioides* (lemonwood)
Harts Creek – The Lake Road	5	600	5.6	*G. littoralis, C. toetoe, C. secta, Phormium tenax* (flax), *C. australis, P. regius, K. ericoides, P. tenuifolium, P. eugenioides, S. microphylla, Coprosma* spp. (coprosma)
Harts Creek – Lochheads Road	20+	600	8.5*	*Nothofagus fusca* (red beech), *Nothofagus solandri* (mountain beech), *P. eugenioides, G. littoralis, P. tenax, C. secta, C. australis, Eucalyptus* spp. (gum tree)
Birdlings Brook	4	1,000	3	*C. toetoe, C. australis, P. tenifolium, Carex secta, K. ericoides, P. regius, P. tenax, Coprosma* spp.

* Only planted on true right hand bank

A paired control site was established upstream for each planted riparian area. The four paired sites are Boggy Creek at Leeston Road (Figure 3.6, site 2 Figure 3.5), Harts Creek at The Lake Road (Figure 3.7, site 4 Figure 3.5), Harts Creek upstream of Lochheads Road (Figure 3.8, site 6 Figure 3.5) and Birdlings Brook at Feredays Road (Figure 3.9, site 8 Figure 3.5). The control sites had not been planted with native vegetation. The control reaches at Boggy Creek and Birdlings Brook were fenced, but at Harts Creek – The Lake Road and Harts Creek – Lochheads Road fences were not in place. At Harts Creek – Lochheads Road the area adjacent to the stream was not actively grazed during the sampling period, while at Harts Creek – The Lake Road the paddock was actively grazed by cattle who were able to access the stream bed until a fence was constructed mid way through the sampling period in June (Figure 3.7, Figure 3.10).

Water samples were collected fortnightly for ten dates during the sampling period of late March to August 2010. Each downstream buffered reach was sampled, followed by the upstream paired control to minimise variation within pairs. Sampling was begun at the same time of day each fortnight and sites were visited in the same order to reduce diurnal variation between dates. Aquatic invertebrate samples were collected monthly at each site. Physical buffer characteristics were measured once during the study period.

Figure 3.6: Boggy Creek control site (paired reach 1).

Figure 3.7: Harts Creek at The Lake Road control site (paired reach 2) in June, after a fence to exclude stock was constructed.

Figure 3.8: Harts Creek control site upstream of Lochheads Road (paired reach 3).

Figure 3.9: Birdlings Brook control site (paired reach 4). Photo courtesy of Hamish Rennie.

Figure 3.10: Cow in-stream at Harts Creek The Lake Road control site in March 2011.

3.3 Characterising sites

Sites were characterised using protocol P2 as outlined by Harding et al. (2009). At each site a GPS reference was taken of the location, these can be found in Appendix 3. In the 50m stretch upstream from the sample site, habitats present were recorded including rapid, riffle, pool, run and backwater. River wetted width was measured with a 30 metre measuring tape, while river depth and sediment depth were measured with a meter staff.

In-stream habitat percentage cover of bed substrate size was recorded in five classes: bed rock (continuous), boulder (> 256 mm), cobble (64 – 255 mm), gravel (2 – 63 mm), and silt/sand/mud (< 2 mm). Substrate embeddedness (percentage of fine sediment surrounding substrate) was recorded. Bed compactness was ranked from: 1 = loose/easily moved, 2 = mostly loose with a little compaction, 3 = moderately packed, 4 = tightly packed. A percentage cover of aquatic macrophytes was also assigned.

On both sides of the bank, buffer width and plant species composition were assessed. Plant density was ranked from: 1 = buffer absent, 2 = 50 – 99 % gaps, 3 = 20 – 50 % gaps, 4 = 1 – 20 % gaps, 5 = completely intact. Bank stability was ranked from: 1 = very low, 2 = low, 3 = moderate, 4 = high, 5 = very high. Shading of the water was estimated and assigned a category from: 1 = little or no shading, 2 = 10 – 25 % shading, 3 = 25 – 50 %, 4 = 50 – 80 % and 5 = > 80 %. The ability of livestock to access to the stream and adjacent land cover/use was noted.

3.4 Water quality sampling

On each sampling date, at the downstream end of each 50 m study stretch, a one litre water sample was taken. Water temperature, conductivity and dissolved oxygen were measured in stream using a HACH HQ40d meter with attached CDC401 conductivity and luminescent dissolved oxygen probes.

The water sample was returned to the Soil and Water Laboratory at Lincoln University where turbidity, soluble phosphates, soluble nitrates, and pH were measured and microbiological plates were prepared. pH was measured using a pH 7.0 calibrated Shindengen ISFET probe KS701. Water turbidity was measured using a light meter MERCK turbiquant 1000IR.

Soluble phosphate levels were calculated using the ascorbic acid method HACH8048 (Hach Limited, 2003). A 10 ml glass bottle was filled with the water sample and a PhosVer3 phosphate powder pillow was added. Following powder pillow addition the sample was shaken for fifteen seconds and left to sit for two minutes. Over this standing period the reaction takes place, and a blue colour indicates the presence of phosphates in the sample. A second bottle was filled with 10 ml of the sample and used to zero the HACH meter. The first bottle was then placed in the meter and the phosphate value was calculated (Hach Limited, 2003).

The cadmium reduction method HACH8171 was used to calculate soluble nitrate levels (Hach Limited, 2003). A 10 ml glass bottle was filled with the water sample and a NitraVer5 powder pillow was added. Following powder pillow addition the sample was shaken vigorously for one minute and left to sit for five minutes. Over this standing period the reaction takes place, and an amber colour indicates nitrates present in the sample. A second bottle was filled with 10 ml of the sample and used to zero the HACH meter. The first bottle was then placed in the meter and the phosphate value was calculated (Hach Limited, 2003).

The membrane filter technique (Merck chromocult coliformagan) was used to prepare microbiological plates. First, a control plate was prepared. A sterilised 47 mm diameter round of 0.45 µm filter paper was placed on a sterilised filter funnel using sterilised tweezers. A small quantity of deionised water was then put onto the filter paper. The vacuum pump was then used to suck the water through the filter paper. The filter paper was then placed onto a prepared agar plate and labelled control.

This process was repeated twice with each sample collected, with either 1 mL or 5 mL of the sample added after the deionised water was put on to ensure the bacteria were spread evenly on the plate. Both 1 mL and 5 mL plates were prepared in order to give quantitative results, producing a total of 17 plates. The plates were then incubated for between 22 and 24 hours, when the number of colony forming units (CFUs) of bacteria of coliforms (red dots), *E.coli* (blue dots), and salmonella (green dots) on each piece of filter paper were counted. The number of CFUs per 100 mL was then calculated by adding the number of CFUs of each form of bacteria found on each of the two volume samples, dividing by six and multiplying by 100.

Standards and calibrations were carried out on equipment used prior to sampling being undertaken. These are important to increase confidence in the data gathered.

3.5 Aquatic invertebrate sampling

Invertebrate samples were collected following semi-quantitative sampling protocols established by Stark et al. (2001) depending on the stream bed condition by the following protocols:

Protocol C1 – hard-bottomed, semi-quantitative sampling was used at sites with gravely bottoms. The sampler wore waders and a standard triangular-frame net (300x300x300 mm with a pore size of 250 µm) was used. An area of habitat was chosen and the net was placed slightly downstream of the sampler. The substrate immediately upstream of the net was disturbed through kicking within 0.5 m from the net mouth. This was repeated at two more areas of riffle habitat to include 0.6 – 1.0 m^2 of streambed at each site (Stark et al., 2001).

Protocol C2 – soft-bottomed, semi-quantitative was used in areas where gravel was not present on the stream bed. This includes areas that were dominated by silt or aquatic macrophytes. The sampler wore waders and a standard triangular-frame net (300 x 300 x 300 mm with a pore size of 250 µm) was used. The net was used to sweep into submerged aquatic plants for a distance of approximately one meter to free organisms, followed by two cleaning sweeps to gather the loose organisms. This was repeated in 3 planted areas to give approximately 0.9 m^2 of streambed at each site (Stark et al., 2001).

In both protocols, organisms were then washed or picked off the net and placed into a shallow white tray with a little water. Tweezers, droppers and a small paintbrush were then used to transfer invertebrates to a sample container with ethanol. The sample container was labelled and returned to the lab for invertebrate identification using keys by Winterbourn et al. (2006) and Otago Regional Council (1997).

After invertebrates were identified, the species richness, percentage EPT taxa and MCI were calculated.

3.6 Flow

Mean daily flow data for Harts Creek at Timber Yard Road for the study period was obtained from Environment Canterbury. Flow data was not available for the other sites, however given they are connected and located in a small geographic area flow was characterised and applied to all sites.

Dates on which data were collected were characterised into medium flow (within one standard deviation of the average flow), low flow (below one standard deviation of the average) and high flow (above one standard deviation of the average flow). Eight sampling dates fitted into the medium flow category, two in the high and none in the low flow category.

3.7 Limitations of field methods

There are some limitations associated with the field methods used. Human error associated with sample measurement, counting microbiological plates and identifying aquatic invertebrates is inevitable. Samplers entering and disturbing the water to collect samples could potentially have affected some parameters. However, the impact of this was minimised by sampling the downstream sites prior to upstream sites.

The ability to gain permission to access private property affected site placement. Therefore control sites may not have been located be directly upstream of the buffered reach, and buffered sites may not be at the downstream end of the planting. This was not deemed to be a significant issue, or gaining access to further areas would have been pursued.

An attempt was made to obtain information of other water quality sampling within the Lake Ellesmere catchment, however this could not be obtained. Flow data was only available from Environment Canterbury at one site, flow on sampling dates was then characterised into low, medium and high and applied to all sampling sites. This method was deemed to be appropriate, given the close geographic location and connectedness of sites.

In terms of the nutrient testing, total nitrate and phosphate tests were not undertaken due to the limited budget of this study. The phosphate samples were not filtered prior to the ascorbic acid method being undertaken. This could mean that the dissolved reactive component may be slightly lower in reality, due to the elevation of this by breakdown of organic matter, however in the majority of cases turbidity was low enough to rule out this concern. The cadmium reduction method for determining nitrates can produce variable accuracy in results when samples are repeatedly tested.

Finally, the limited time frame of this thesis only allowed sampling to be undertaken fortnightly over ten dates autumn and winter, and therefore results are not representative of the year round patterns.

The limitations identified above are consistent with other water quality studies, and generally accepted as acceptable sources of error. This study aimed to limit these sources of error and aimed to ensure data consistency through undertaking sampling on multiple dates.

3.8 Data analysis methods

Initial data analyses were carried out using GenStat 12.2 (VSN International Ltd, 2010). Data were run through a multivariate principal components analysis (PCA) to establish the source of variation in the dependent variables; pH, temperature, conductivity, turbidity, dissolved oxygen, soluble phosphate, soluble nitrate and bacterial counts. Results of the PCA suggested that river flow had a large influence on the dependent variables, potentially masking the any effects of the independent variable; presence/absence of a riparian buffer. Data collected on the two dates with high flows was excluded from further analysis. The remaining data was averaged over the sampling dates to reduce temporal variation in dependent variables. A final PCA was then undertaken with the effects of flow and date removed.

Data were checked for normality by inspection of probability plots and log transformed where appropriate before a series of restricted maximum likelihood (REML) tests were carried out for all dependent variables. Where the REML output indicated significant differences, pairwise comparisons of means using Fishers least significant difference (LSD) tests ($\alpha = 0.05$) (Zar, 2009) were undertaken to determine which sites had significant differences between treatments (control or buffer).

A principle coordinate analysis using Steinhaus distance matrix (Legendre & Legendre, 1998) was used to produce a two-dimensional species ordination plot showing the variation in invertebrate community composition across all sample sites. This was carried out in R version 2.11.1 (The R Foundation for Statistical Computing, 2010).

Chapter 4

Results

4.1 Introduction

In this section, the results of fieldwork and statistical analysis will be presented. Firstly, characterisations of the paired control and buffer sites are detailed. This is followed by results of water quality assessments, bacterial concentrations and macroinvertebrate communities.

Normally only significant results would be presented in a results section, however in this study a statistically non-significant result still represents a result that is of significance in terms of the effectiveness of riparian buffer at protecting water quality. Because of this, in this thesis all results statically significant or not have been presented.

In terms of the results presented, treatment refers to whether the site had been restored or not. A treatment effect means that results are significantly different between the control and buffered reaches. A site effect means that results are significantly different between the four paired sites sampled. An interaction between treatment and site means that the effects of treatment differed between the sampling sites.

Raw data are attached in Appendix 5 and 6. Flow characterisations are attached in Appendix 7.

4.2 Site characteristics

Channel widths were found to be similar between buffer and control pairs (Table 4.1). Channel depth was much deeper at Harts Creek – The Lake Road site (1.1 m) compared with the other seven sites (range from 0.21 – 0.57 m) (Table 4.1). Bed substrate was gravelly at the Boggy Creek pair, Harts Creek – Lochheads Road control and Birdlings Brook buffered sites, cobbles at Harts Creek – The Lake Road control and Lochheads Road buffer and silt at Harts Creek – The Lake Road buffer and Birdlings Brook control (substrate sizes as defined in Methodology section 3.3 – Characterising sites) (Table 4.1). Aquatic macrophytes were present at all sites except for Harts Creek – Lochheads Road control (Table 4.1). The sites with planted riparian vegetation were not fully shaded, however more shading was observed at the buffered sites at Boggy Creek, Harts Creek – The Lake Road and Birdlings Brook (Table

4.1). At Harts Creek – Lochheads Road control several large *Salix fragilis* (willows) on the true left provided noteworthy shading to the channel (Table 4.1).

The width of planted buffer was 3 m on either side at Birdlings Brook, 5 m on either side at Boggy Creek, 5.6 m either side at Harts Creek – The Lake Road and 8.5 m on the true right hand bank at Harts Creek – Lochheads Road (Table 3.1). The age of planted buffers ranged from two years (Boggy Creek) to more than 20 (Harts Creek – The Lake Road) (Table 3.1). Three of the buffered sites are fenced to exclude stock from the waterway, and the fourth site is not grazed (Harts Creek at Lochheads Road). The control reaches on Boggy Creek and Birdlings Brook were fenced, but at Harts Creek – The Lake Road and Harts Creek – Lochheads Road fences were not in place. At Harts Creek – Lochheads Road the area adjacent to the stream was not actively grazed during the sampling period, while at Harts Creek – The Lake Road the paddock was actively grazed by cattle who were able to access the stream bed until a fence was constructed mid way through the sampling period in June (Figure 3.7, Figure 3.10).

The predominant land use in the study area is agriculture. The Boggy Creek and Birdlings Brook buffered sites are grazed by dairy cows, the Harts Creek – The Lake Road pair and Birdlings Brook control sites are grazed by beef cows, the Harts Creek – Lochheads Road pair are grazed by sheep and used for organic vegetable farming and the Boggy Creek control site is in a rural residential area. Aerial photos of the study area and paired site locations are attached in Appendix 4 to give an idea of the surrounding land uses.

Table 4.1: Site Habitat characteristics.

	Boggy Creek		**Harts Creek – The Lake Road**		**Harts Creek – Lochheads Road**		**Birdlings Brook**	
	Buffer	**Control**	**Buffer**	**Control**	**Buffer**	**Control**	**Buffer**	**Control**
Habitats present in 50 m stretch	Riffle	Riffle	Pool	Rapid, run, riffle	Rapid and run	Run, rapid, backwater	Run, riffle	Run
Channel width (m)	4	2.6	11	10	5.1	5.9	3	4.3
Channel depth (m)	0.29	0.21	1.1	0.46	0.32	0.57	0.3	0.4
Bed substrate (dominant, others)	Gravel (silt, cobbles)	Gravel (cobbles)	Silt	Cobbles (gravel, boulder)	Cobbles (gravel, boulder)	Gravel (cobbles, silt)	Gravel (cobbles, silt)	Silt (gravel, cobbles)
Substrate embeddedness (% sediment surrounding	10	0	All sediment	20	10	0	0	15

substrate)								
Substrate compactness	Mostly loose	Mostly loose	Loose / easily moved	Mostly loose	Mostly loose	Mostly loose	Mostly loose	Mostly loose
Macrophytes (% cover summer, winter)	50, 20	50, 10	60, 10	50, 40	60, 40	0, 0	70, 20	85, 85
Dominant macrophyte type	Monkey musk	Monkey musk	Elodea	Monkey musk	Monkey musk	None	Monkey musk	Monkey musk
Shading (%)	10-25	Little or no shading	10-25	Little or no shading	10-25	25-50	25-50	Little or no shading
Buffer intactness (% gaps)	1-20	N/A	1-20	N/A	1-20 on TR, 50-99 on TL	N/A	1-20	N/A
Bank stability	Very high	Very high	Very high	High	Very high	Very high	Very High	Very high

4.3 Water quality sampling

A multivariate principle components analysis (PCA) of all data was undertaken to establish the source of variation in the dependent variables (pH, temperature, conductivity, turbidity, dissolved oxygen, soluble phosphate, soluble nitrate and bacterial counts). The analysis showed that 70 % of the percentage variation in dependent variables can be explained in three dimensions (Table 4.2).

Table 4.2: Percentage variation in dependent variables explained by multivariate PCA.

Dimension	**1**	**2**	**3**
Percentage variation	39.84	18.53	11.18

The PCA showed that the flow rate massively influenced the dependent variables, potentially masking the treatment effects (Figure 4.1). As the data collected on high flow dates was an order of magnitude higher, this made it difficult to further analyse the data in terms of treatment effects (differences between control and buffered reaches). Because of this, the data collected on the two dates with high flow were excluded from further analysis.

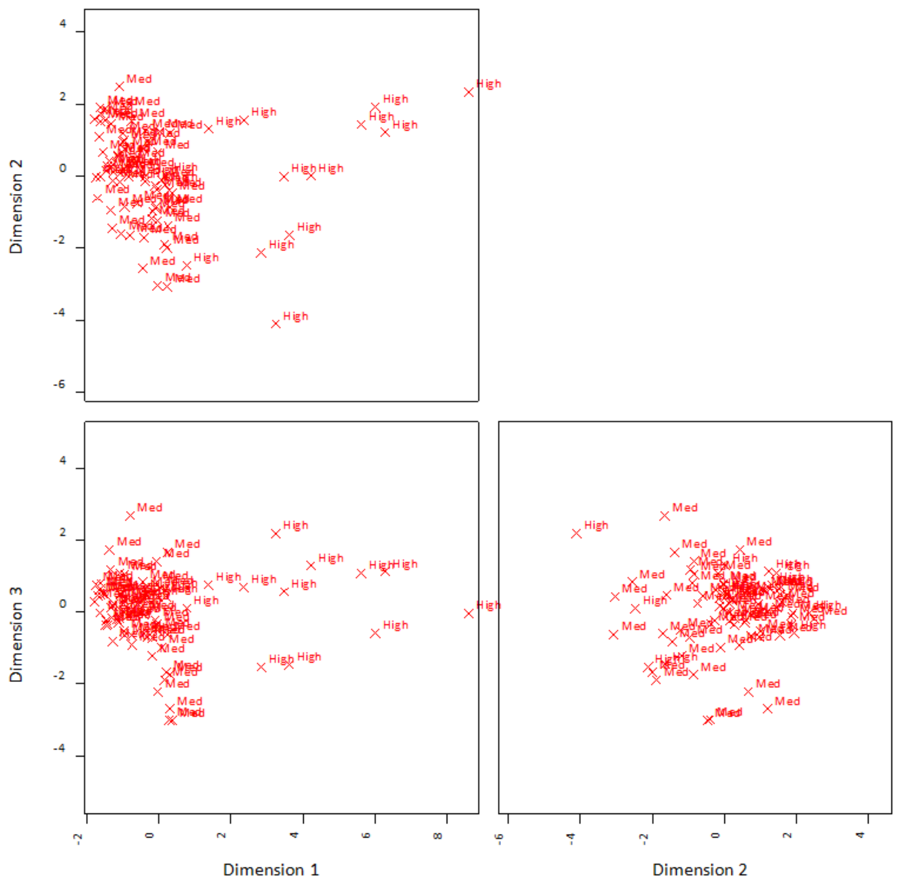

Figure 4.1: Three-dimensional multivariate PCA output showing spread of dependent variables, labelled by flow rate (medium or high).

The multivariate PCA excluding high flow data also explained 70 % of variation seen between sites (42 % on axis one and 28 % on axis two (Figure 4.2). With the effects of high flow on dependent variables excluded, shifts between control and buffer reaches were evident. However these differences were mixed between the four site pairs (Figure 4.2).

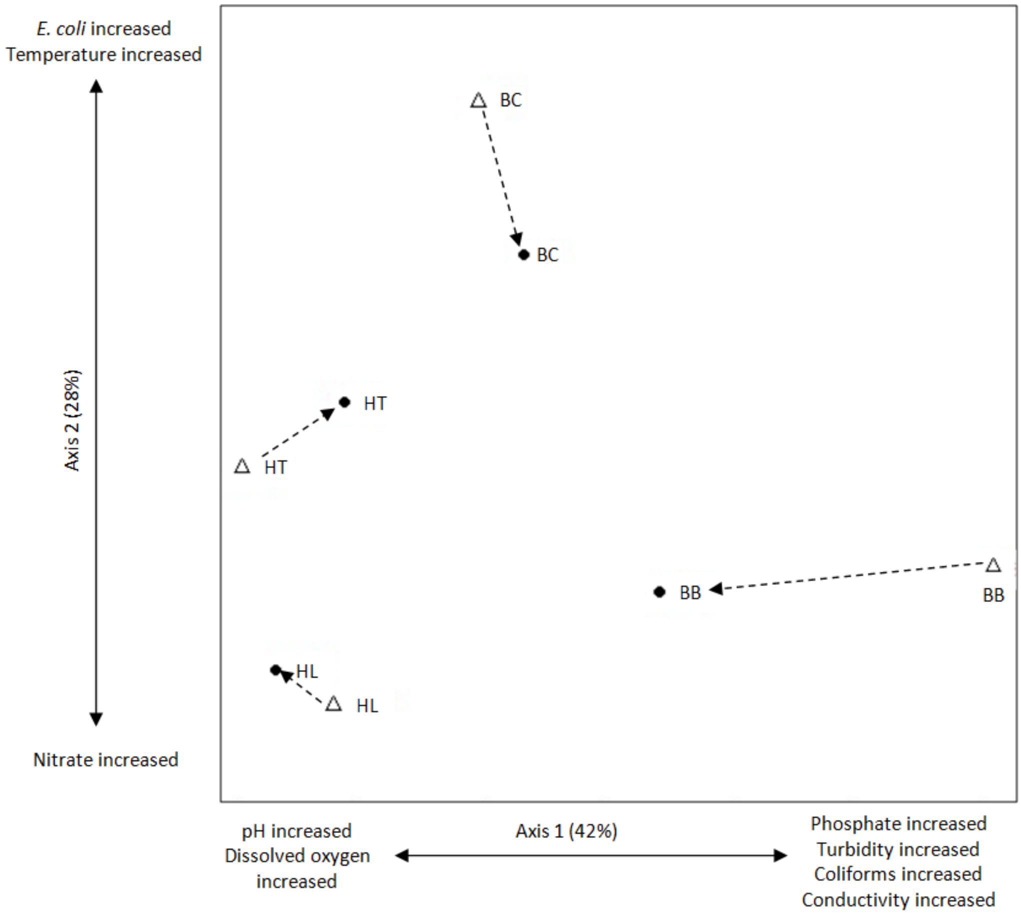

Figure 4.2: Two-dimensional multivariate PCA output showing spread of dependent variables at buffer (closed circles) and control (open triangles) sites on medium flow dates. Arrows indicate the direction of change between the control and buffer reach at each pair. Abbreviations of site code are BB – Birdlings Brook, BC – Boggy Creek, HL – Harts Creek at Lochheads Road, HT – Harts Creek at The Lake Road.

4.3.1 pH

The mean pH over all sites on all dates was 7. The pH was found to range between 6.3 and 7.4 over the sampling dates (Table 4.3).

Table 4.3: Minimum, mean and maximum pH values recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
pH	Minimum	6.3	6.6	6.4	6.7	6.5	6.6	6.6	6.8
	Mean	6.84	6.92	7.04	7.04	7.08	7.01	7	7.03
	Maximum	7.2	7.2	7.4	7.4	7.4	7.2	7.4	7.4

Note: includes dates with high flows.

There was no significant main effect of treatment (x^2 = 0.00, DF = 1, p = 1.000) or site (x^2 = 2.05, DF = 3, p = 0.569) on pH. There was also no significant interaction between site and treatment (x^2 = 2.02, DF = 3, p = 0.575) (Figure 4.3).

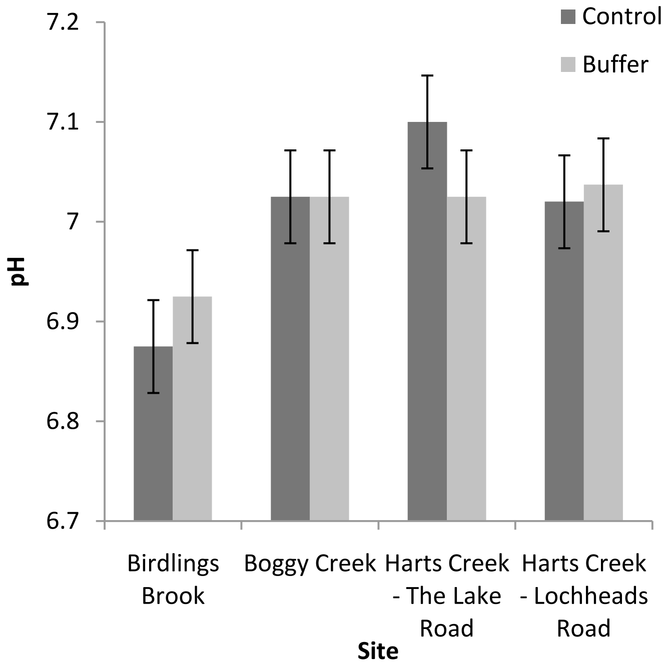

Figure 4.3: Average pH levels (±SEM) between control and buffer sites.

4.3.2 Water temperature

The mean water temperature over all sites on all dates was 10.9 ^{o}C. Water temperature ranged from 5.1 to 15.7 ^{o}C over the sampling dates (Table 4.4).

Table 4.4: Minimum, mean and maximum temperatures recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Temperature ^{o}C	Minimum	8.2	8.4	5.7	5.1	8.2	8.2	9.8	9.7
	Mean	11.48	11.07	9.55	9.53	11.05	11.1	11.77	11.74
	Maximum	15.7	13.3	13.2	13.3	12.6	12.8	13.4	13.4

Note: includes dates with high flows.

There was no significant main effect of treatment (x^2 = 1.49, DF = 1, p = 0.232) or site (x^2 = 5.71, DF = 3, p = 0.152) on water temperature. Again there was no significant interaction between site and treatment (x^2 = 6.92, DF = 3, p = 0.098) (Figure 4.4).

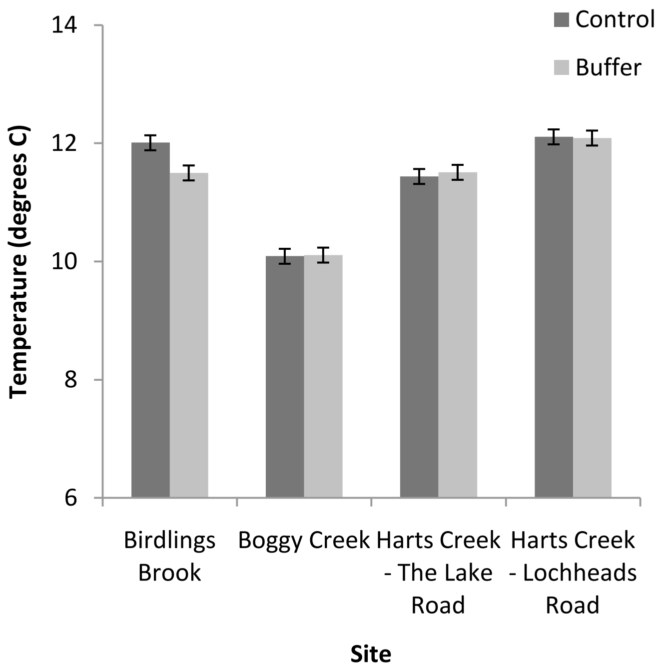

Figure 4.4: Average temperature (±SEM) between control and buffer sites.

4.3.3 Conductivity

The mean conductivity over all sites on all dates was 298 µS/cm. Conductivity ranged from 198.4 – 404 µS/cm over the sampling period (Table 4.5).

Table 4.5: Minimum, mean and maximum conductivity recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Conductivity µS/cm	Minimum	292	311	262	246	200.5	198.4	239	240
	Mean	339.8	347.1	330.6	330.6	236.34	235.7	245.9	246.1
	Maximum	414	400	401	409	276	276	251	250

Note: includes dates with high flows.

There was found to be a significant main effect of treatment, (x^2 = 8.17, DF = 1, p = 0.008) where conductivity was higher in planted areas. There was also found to be a significant effect of site (x^2 = 60.60, DF = 3, p < 0.001) and a significant interaction between treatment and site (x^2 = 13.67, DF = 3, p = 0.01) (Figure 4.5). Post hoc pairwise comparisons using Fishers LSD tests (α = 0.05) indicated that only the Birdlings Brook pair had a significant difference between the planted and unplanted treatment (p < 0.05), where conductivity was higher at the buffered site. There were no significant differences between the planted and non-planted treatments at all other sites (p > 0.05) (Figure 4.5).

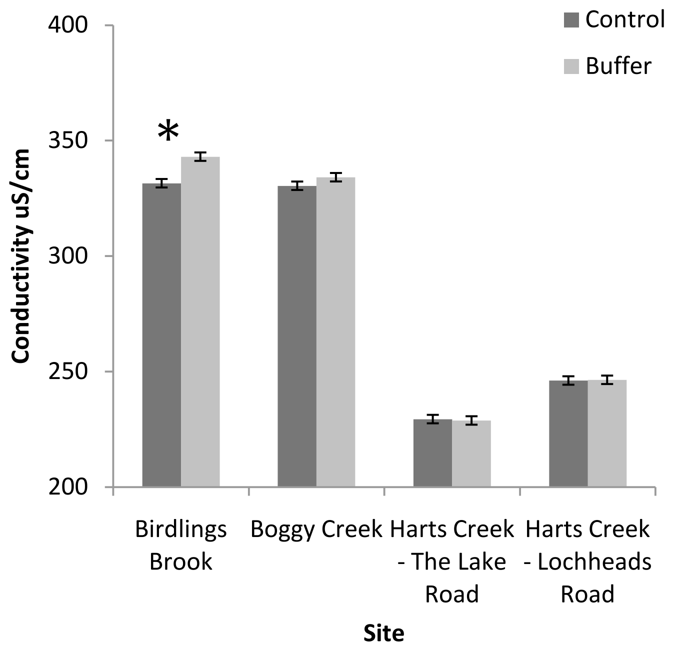

Figure 4.5: Average conductivity (±SEM) between control and buffer sites.
Significant differences for each pair of control and buffer sites are marked with asterisks (* LSD test p < 0.05).

4.3.4 Turbidity

The mean turbidity over all sites on all dates was 16.8 NTUs. Turbidity was found to range from 0.33 – 96.07 NTUs over the sampling dates (Table 4.6).

Table 4.6: Minimum, mean and maximum turbidity values recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Turbidity (NTUs)	Minimum	2.67	0.33	1.36	0.84	2.12	1.82	0.69	0.84
	Mean	24.612	18.007	19.214	20.687	17.264	15.999	9.5	9.406
	Maximum	91.81	73.87	67.53	96.07	73.36	72.04	44.41	42.6

Note: includes dates with high flows.

There was found to be a significant main effect of treatment, (x^2 = 4.45, DF = 1, p = 0.044), where turbidity was lower in planted areas. There was no effect of site (x^2 = 1.75, DF = 3, p = 0.630) or interaction between site and treatment (x^2 = 4.06, DF = 3, p = 0.277) (Figure 4.6).

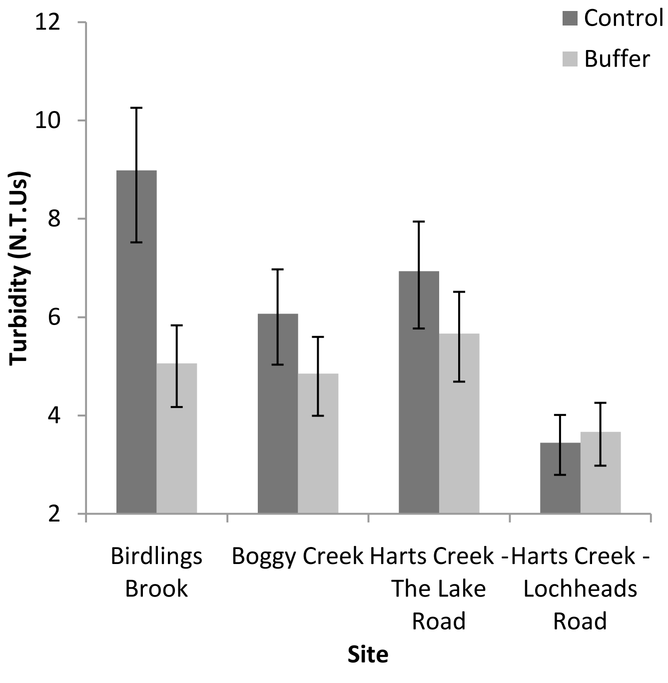

Figure 4.6: Average water turbidity (±SEM) between control and buffer sites.

4.3.5 Dissolved oxygen

The mean dissolved oxygen over all sites on all dates was 85.1 % saturation. Dissolved oxygen was found to range from 51.1 – 100.5 % saturation over the sampling dates (Table 4.7).

Table 4.7: Minimum, mean and maximum dissolved oxygen values recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Dissolved oxygen % sat	Minimum	51.1	70.4	80.3	84.8	81.4	82.2	82.9	83.9
	Mean	67.8	79.23	89.08	91.7	88.26	90.13	86.77	87.98
	Maximum	76.1	93.6	100	97.6	96	100.5	91.1	91.9

Note: includes dates with high flows.

There was found to be a significant main effect of treatment, (x^2 = 25.11, DF = 1, p < 0.001) where dissolved oxygen was higher in areas with planted buffers. There was also found to be a significant effect of site (x^2 = 68.65, DF = 3, p < 0.001) and a significant interaction between treatment and site (x^2 = 25.14, DF = 3, p < 0.001) (Figure 4.7). Post hoc pairwise comparisons using Fishers LSD tests (α = 0.05) indicated that only the Birdlings Brook pair had a significant difference in dissolved oxygen levels between the planted and unplanted treatment (p < 0.05), where dissolved oxygen was higher at the buffered site. There were no significant differences between the planted and non-planted treatments at all other sites (p > 0.05) (Figure 4.7).

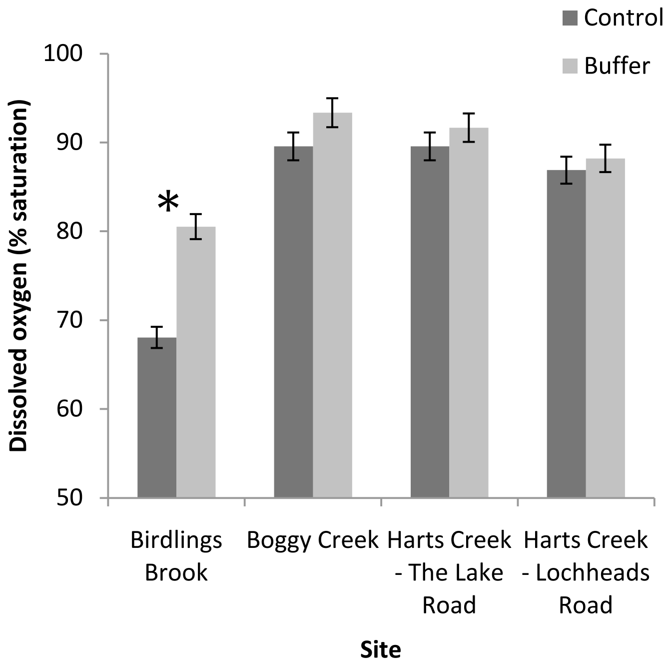

Figure 4.7: Average dissolved oxygen saturation (±SEM) between control and buffer sites. Significant differences for each pair of control and buffer sites are marked with asterisks (* LSD test $p < 0.05$).

4.3.6 Soluble Phosphate

The mean phosphate concentration over all sites on all dates was 0.33 mg/L. Phosphate levels were found to range from 0.05 – 1.18 mg/L over the sampling dates (Table 4.8).

Table 4.8: Minimum, mean and maximum soluble phosphate values recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Soluble phosphates mg/L	Minimum	0.1	0.13	0.17	0.15	0.1	0.14	0.05	0.08
	Mean	0.423	0.351	0.413	0.535	0.211	0.311	0.197	0.229
	Maximum	0.76	0.8	1.16	1.18	0.48	0.66	0.35	0.38

Note: includes dates with high flows.

There was no significant main effect of treatment (x^2 = 1.76, DF = 1, p = 0.196) on soluble phosphate levels. However, a significant difference in phosphate levels was seen between sites (x^2 = 9.35, DF = 3, p = 0.042). There was no significant interaction between treatment and site (x^2 = 6.09, DF = 3, p = 0.133) (Figure 4.8).

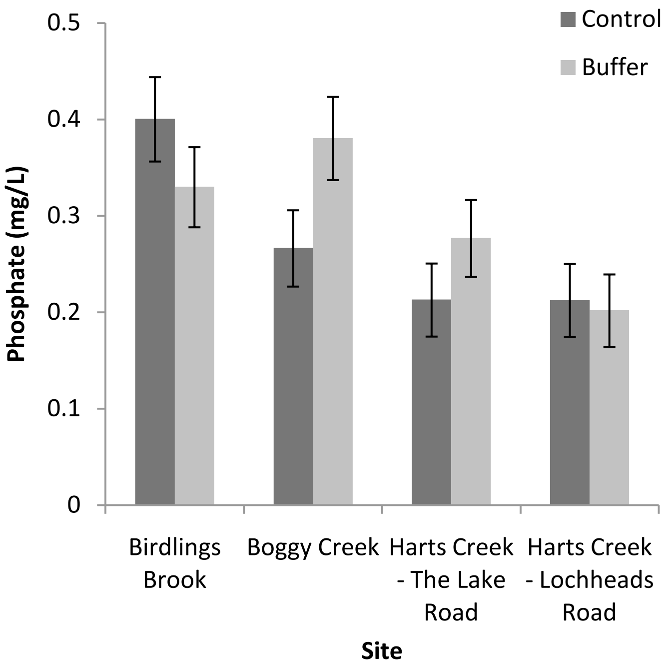

Figure 4.8: Average phosphate levels (±SEM) between control and buffer sites.

4.3.7 Soluble Nitrate

The mean nitrate concentration over all sites on all dates was 3.35 mg/L. Nitrate levels were found to range from 1 – 5.8 mg/L over the sampling dates (Table 4.9).

Table 4.9: Minimum, mean and maximum soluble nitrate values recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Soluble nitrates mg/L	Minimum	2.1	2	1	1.2	2.2	2.2	2.8	3
	Mean	3.3	3.68	2.74	3.11	2.8	3.26	4.04	3.86
	Maximum	4.2	5.8	4.1	3.8	3.6	4.9	4.7	5

Note: includes dates with high flows.

There was no significant main effect of treatment (x^2 = 3.78, DF = 1, p = 0.062) on soluble nitrate levels. However, a significant difference in nitrate levels was seen between sites (x^2 = 10.60, DF = 3, p = 0.027). There was no significant interaction between treatment and site (x^2 = 2.70, DF = 3, p = 0.453) (Figure 4.9).

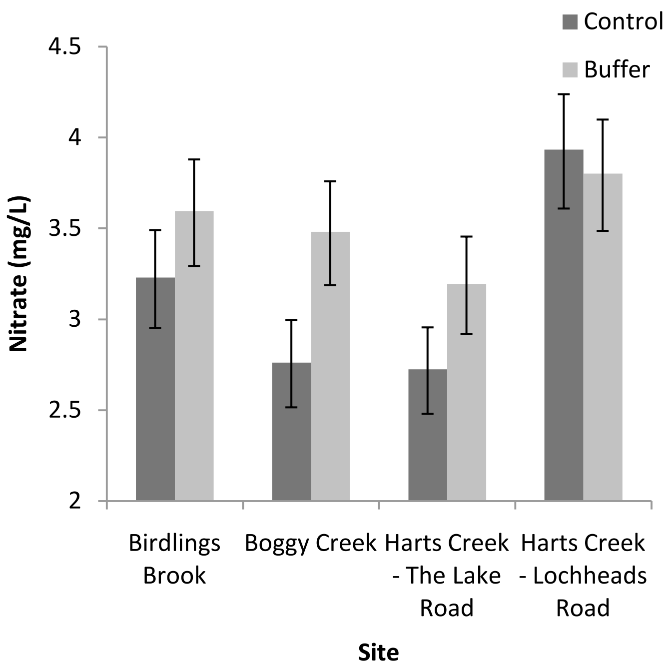

Figure 4.9: Average nitrate levels (±SEM) between control and buffer sites.

4.4 Bacterial sampling

4.4.1 Coliforms

The mean concentration of coliforms over all sites on all dates was 2149.9 CFUs/100 mL. Coliform levels were found to range from 166.7 – 12466.7 CFUs/100 mL over the sampling dates (Table 4.10).

Table 4.10: Minimum, mean and maximum coliform levels recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Coliforms CFUs/100 mL	Minimum	300	366.7	366.7	583.3	233.3	283.3	166.7	300
	Mean	2604.8	2087.7	2082	2243.9	2103.7	2471.2	1896	1709
	Maximum	8266.7	9333.3	7600	8600	12267	12467	8000	8200

Note: includes dates with high flows.

There was no main effect of treatment (x^2 = 0.55, DF = 1, p = 0.466) or site (x^2 = 5.54, DF = 3, p = 0.161) on coliform levels. There was also no significant interaction between site and treatment (x^2 = 5.71, DF = 3, p = 0.152) (Figure 4.10).

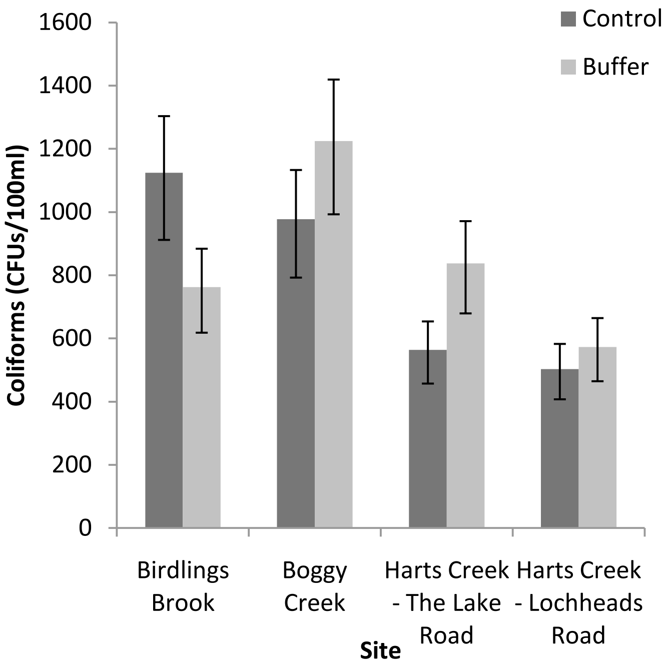

Figure 4.10: Average coliforms (±SEM) between control and buffer sites.

4.4.2 E. coli

The mean concentration of *E. coli* over all sites on all dates was 1719.6 CFUs/100 mL. *E. coli* levels ranged from 66.7 – 22733.3 CFUs/100 mL over the sampling period (Table 4.11).

Table 4.11: Minimum, mean and maximum *E. coli* levels recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
E.coli CFUs/100mL	Minimum	66.7	66.7	250	240	183.3	150	100	100
	Mean	948.2	810.6	3582	3521.7	1607.6	1544.8	815.4	926.5
	Maximum	6066.7	4066.7	14466.7	22733.3	9933.3	10666.7	5283.3	5366.7

Note: includes dates with high flows.

There was no significant main effect of treatment on *E. coli* levels (x^2 = 2.83, DF = 1, p = 0.104). However, there was found to be a significant effect of site (x^2 = 38.91, DF = 3, p < 0.001) and a significant interaction between treatment and site (x^2 = 16.71, DF = 3, p = 0.004) (Figure 4.11). Post hoc pairwise comparisons using Fishers LSD tests (α = 0.05) indicated that only the Boggy Creek pair had a significant difference in *E. coli* levels between the planted and unplanted treatment (p < 0.05), where *E. coli* levels were lower at the buffered site. There were no significant differences between the planted and non-planted treatments at all other sites (p > 0.05) (Figure 4.11).

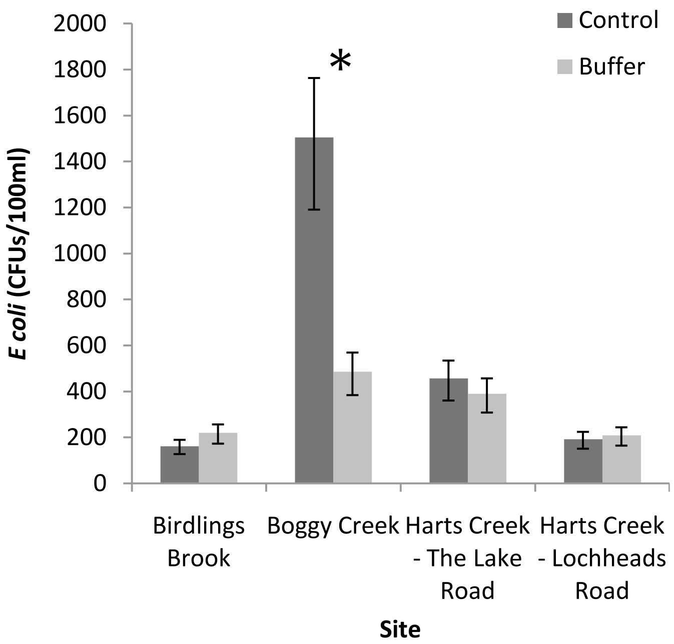

Figure 4.11: Average *E. coli* (±SEM) between control and buffer sites.
Significant differences for each pair of control and buffer sites are marked with asterisks (* p < 0.05).

4.4.3 Salmonella

The mean salmonella level over all sites on all dates was 172 CFUs/100 mL. Salmonella levels were found to range from 0 – 1333.3 CFUs/100 mL over the sampling dates (Table 4.12).

Table 4.12: Minimum, mean and maximum salmonella levels recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Salmonella CFUs/100 ml	Minimum	0	0	0	18.2	16.7	16.7	0	0
	Mean	173.5	124	144.6	213.8	145.5	261.6	129.7	183.8
	Maximum	1200	866.7	733.3	1200	1000	1333.3	800	1133.3

Note: includes dates with high flows.

There was no main effect of treatment (x^2 = 3.32, DF = 1, p = 0.079) or site (x^2 = 2.37, DF = 3, p = 0.510) on salmonella levels. There was also no significant interaction between site and treatment (x^2 = 0.66, DF = 3, p = 0.881) (Figure 4.12).

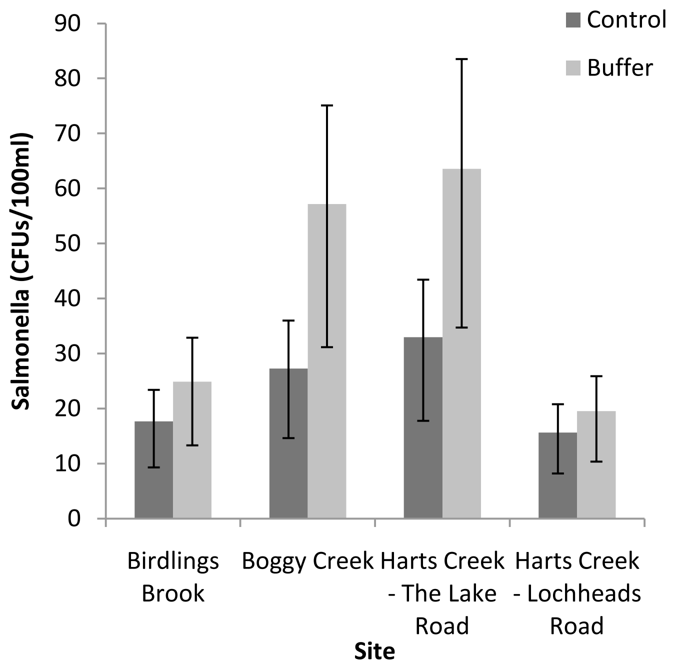

Figure 4.12: Average salmonella levels (±SEM) between control and buffer sites.

4.5 Macroinvertebrate sampling

4.5.1 Species richness

The mean species richness over all sites on all dates was 11. Species richness ranged from 6 – 14 over the sampling dates (Table 4.13).

Table 4.13: Minimum, mean and maximum species richness recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
Species richness	Minimum	6	7	8	7	12	9	8	13
	Mean	8	11.2	9.8	10	13.4	10.8	11.6	13.2
	Maximum	12	13	12	12	14	12	14	14

Note: all invertebrate samples were taken on dates with medium flow.

There was no significant main effect of treatment on species richness (x^2 = 1.00, DF = 1, p = 0.326). However, there was found to be a significant effect of site, (x^2 = 17.55, DF = 3, p = 0.003) and a significant interaction between site and treatment (x^2 = 12.57, DF = 3, p = 0.013) (Figure 4.13).

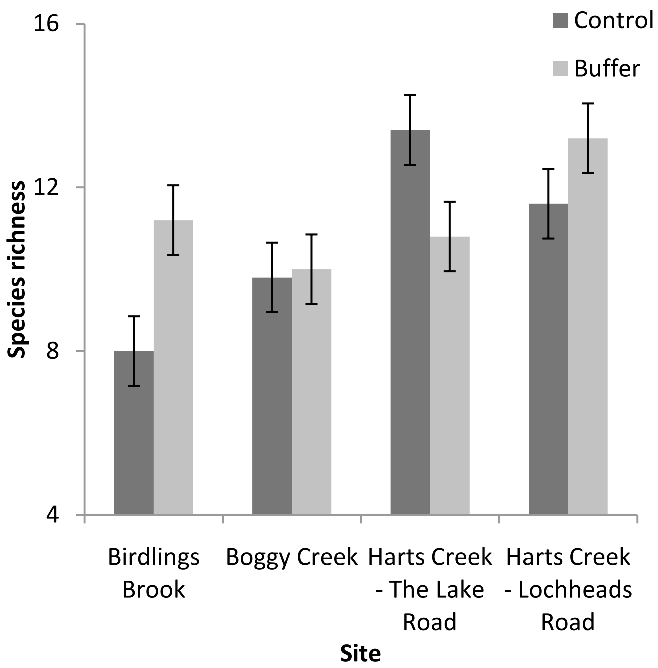

Figure 4.13: Average species richness (±SEM) between control and buffer sites.

The principle coordinate analysis using Steinhaus distance matrix explained 75 % of variation seen between sites (55 % on axis one and 20 % on axis two) (Figure 4.14). Axis one showed a community shift from molluscs (Mollusca) to mayflies (Ephemeroptera) and some caddisflies

(Trichoptera including *Helicopsyche, Hydrobiosis* & *Oligna*), axis two showed a shift from amphipods (Amphipoda), chironomids (Chironomidae) and purse caddisflies (*Oxyethira*) to mayflies (Ephemeroptera) and caddisflies (including *Hudsonema*) (Figure 4.14).

In terms of differences in community composition between control and buffer sites, there appear to have been mixed responses (Figure 4.14). Community composition appears to be driven more by the bed substrate present at the site, than either whether the site was planted or not or time since planting had occurred. This can be seen by the cluster of sites in silt substrate at the left end of axis one (Figure 4.14).

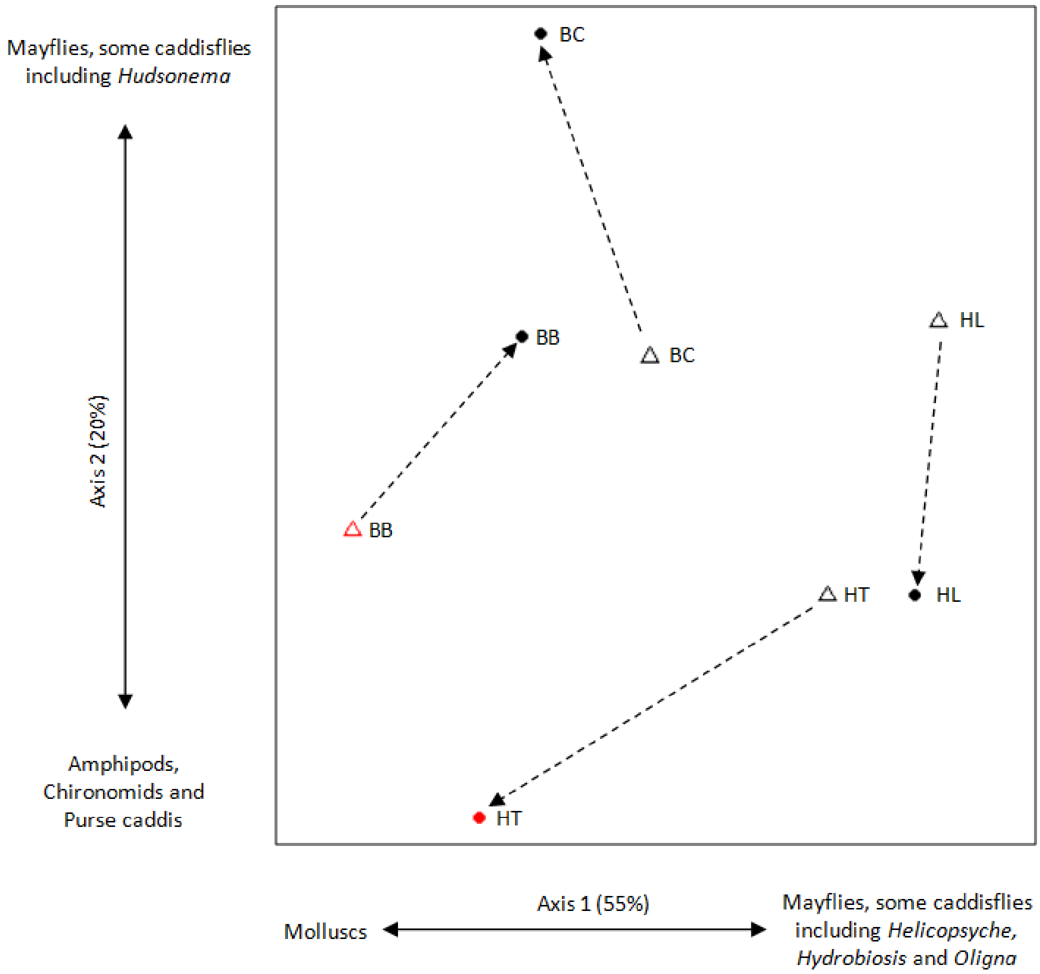

Figure 4.14: Two-dimensional ordination plot of invertebrate community composition at buffer (closed circles) and control (open triangles) sites. Arrows indicate the direction of change between the control and buffer reach at each pair. Red indicates the site has a silt substrate, black indicates gravel or cobbles. Abbreviations of site code are BB – Birdlings Brook, BC – Boggy Creek, HL – Harts Creek at Lochheads Road, HT – Harts Creek at The Lake Road.

4.5.2 Macroinvertebrate community index

The mean MCI value over all sites on all dates was 97.3. MCI values ranged from 57.1 – 127.1 over the sampling dates (Table 4.14).

Table 4.14: Minimum, mean and maximum MCI values recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
MCI score	Minimum	57.1	85.7	90.0	91.7	85.0	74.5	107.5	113.8
	Mean	66.8	92.6	96.1	98.6	105.5	82.3	115.4	120.8
	Maximum	73.3	96.7	115.6	106.7	118.5	89.1	123.3	127.1

Note: all invertebrate samples were taken on dates with medium flow.

There was no significant main effect of treatment on macroinvertebrate community index score (x^2 = 1.17, DF = 1, p = 0.288). However, there was found to be a significant effect of site, (x^2 = 127.27, DF = 3, p < 0.001) and a significant interaction between site and treatment (x^2 = 51.47, DF = 3, p < 0.001) (Figure 4.15). Post hoc pairwise comparisons using Fishers LSD tests (α = 0.05) indicated that there were significant differences in MCI scores at the Birdlings Brook and Harts Creek at the Lake Road pairs (p < 0.05). At the Birdlings Brook pair, the MCI score was significantly lower at the unplanted site, however at Harts Creek the Lake Road pair, the MCI score was significantly lower at the planted site (Figure 4.15). There were no significant differences between the planted and non-planted treatments at the Boggy Creek and Harts Creek at Lochheads Road pairs (p > 0.05) (Figure 4.15).

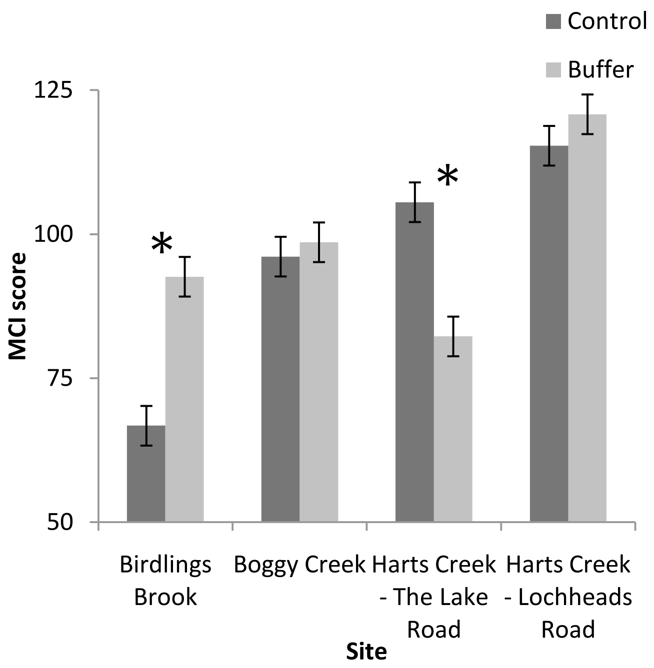

Figure 4.15: Average MCI rating (±SEM) between control and buffer sites. Significant differences for each pair of control and buffer sites are marked with asterisks (* $p < 0.05$).

4.5.3 Percentage Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa

The mean percentage of EPT taxa over all sites on all dates was 41.5 %. Percentages of EPT taxa ranged from 0 – 71.4 % over the sampling dates (Table 4.15).

Table 4.15: Minimum, mean and maximum percentage EPT taxa recorded at each site over the sampling dates.

		Birdlings Brook		Boggy Creek		Harts Creek – The Lake Road		Harts Creek – Lochheads Road	
		Control	Buffer	Control	Buffer	Control	Buffer	Control	Buffer
EPT taxa %	Minimum	0.0	14.3	33.3	40.0	33.3	18.2	53.8	61.5
	Mean	5.6	35.8	43.3	48.9	50.4	24.0	58.9	65.2
	Maximum	16.7	50.0	66.7	57.1	71.4	27.3	66.7	69.2

Note: all invertebrate samples were taken on dates with medium flow.

There was no significant main effect of treatment on percentage Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa (x^2 = 1.51, DF = 1, p = 0.227). However, there was found to be a significant effect of site, (x^2 = 89.33, DF = 3, p < 0.001), and a significant interaction between site and treatment (x^2 = 40.40, DF = 3, p < 0.001) (Figure 4.16). Post hoc pairwise comparisons using Fishers LSD tests (α = 0.05) indicated that there were significant differences in percentage EPT taxa at the Birdlings Brook and Harts Creek at the Lake Road pairs (p < 0.05). At the Birdlings Brook pair, the percentage EPT taxa was significantly lower at the unplanted site, however at Harts Creek the Lake Road pair, the percentage EPT taxa score was significantly lower at the planted site (Figure 4.16). There were no significant differences between the planted and non-planted treatments at the Boggy Creek and Harts Creek at Lochheads Road pairs (p > 0.05) (Figure 4.16).

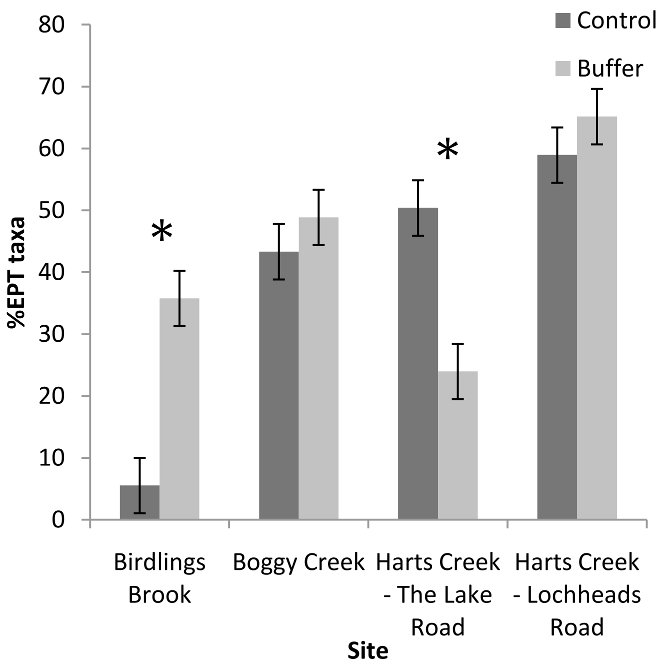

Figure 4.16:Average percentage EPT taxa (±SEM) between control and buffer sites. Significant differences for each pair of control and buffer sites are marked with asterisks (* $p < 0.05$).

Chapter 5
Discussion

5.1 Introduction

The goal of this study was to evaluate the impact of riparian plantings on water quality using a case study in the Lake Ellesmere catchment. A paired reach design was used to compare restored riparian buffers with unplanted control areas upstream. In this chapter, a summary of the results of this study is presented, followed by an evaluation of the effectiveness of riparian planting and a discussion of the factors affecting their success.

5.2 Summary of main findings

5.2.1 Site characterisations

The dominant bed substrate was found to be gravel or cobbles at six of the eight sites, with silt being the dominant substrate at Harts Creek – The Lake Road buffer and Birdlings Brook control (Table 4.1). The sites which had restored riparian buffers had not established full canopy cover over the stream, however more shading was observed at the buffered sites than the control sites at Boggy Creek, Harts Creek – The Lake Road and Birdlings Brook (Table 4.1).

5.2.2 Water quality, bacterial counts and macroinvertebrate communities

The multivariate PCA showed that flow rate had a massive influence on dependent variables, masking the effect of treatment (Figure 4.1). Because of this, the data collected on the two dates with high flow were excluded from further analysis. With the effects of high flow excluded, shifts between control and buffer reaches were evident; however these differences were not consistent between the four site pairs (Figure 4.2).

There were significant main effects of treatment on conductivity, turbidity and dissolved oxygen levels (Figures 4.5, 4.6 & 4.7). Dissolved oxygen and conductivity were found to be significantly higher in areas where planting had occurred. Post hoc pairwise comparisons showed that the least significant differences (LSDs) in both dissolved oxygen and conductivity were only significant at the Birdlings Brook pair (Figures 4.5 & 4.7). Turbidity was found to be significantly lower in areas that had been planted (Figure 4.6). No differences between planted and control areas were seen in pH, water temperature, soluble phosphate, soluble nitrate, coliforms, *E. coli*, salmonella, species richness, macroinvertebrate community index or percentage EPT taxa (Figures 4.3, 4.4, 4.8, 4.9, 4.10, 4.11, 4.12, 4.13, 4.15 & 4.16).

Post hoc pairwise comparisons showed a significant difference in *E. coli* levels only at the Boggy Creek pair, where levels were significantly higher at the unplanted site when compared with the planted pair (Figure 4.11). MCI and percentage EPT taxa values were significantly higher at the Birdlings Brook planted site compared with the unplanted, but significantly lower at Harts Creek – The Lake Road planted site compared with the unplanted (Figures 4.15 & 4.16).

Significant site effects (variation in factor measured between sites) were seen in conductivity, dissolved oxygen, phosphate, nitrate, *E. coli*, species richness, MCI, and percentage EPT taxa (Figures 4.5, 4.7, 4.8, 4.9, 4.11, 4.13, 4.15 & 4.16). Significant interactions between treatment and site (differing effects of treatment depending on site) were seen in conductivity, dissolved oxygen, *E. coli*, species richness, MCI and percentage EPT taxa (Figures 4.5, 4.7, 4.11, 4.13, 4.15 & 4.16).

Mixed responses were seen in invertebrate community composition between control and buffer sites, and community composition appears to be driven more by the bed substrate present at the site, than whether the site was planted or not (Figure 4.14).

5.2.3 Acceptable ranges of water quality variables

In this section, the water quality variables measured in this study are compared with the acceptable ranges set by different authorities and presented in section 2.6 and Table 2.1. The water quality measurements presented in this section include those collected on dates with high flows.

pH ranged from 6.3 to 7.4, falling within the range of 6.0 to 8.5, as recommended by James (1999). The range of values observed at each sampling site were all out of the range of 7.2 to 7.8 as recommended by the ANZECC (2000), and all sites had mean values below 7.2 (Table 4.3). However, as the ANZECC ranges are often considered to be overly stringent (eg. Milne and Perrie 2006), these values are considered to be acceptable falling within the range adopted by James (1999).

Water temperature ranged from 5.1 – 15.7 ^{o}C, well below the limit of 20 ^{o}C (Quinn & Hickey, 1990).

Given that no published guideline for conductivity levels in New Zealand could be found, conductivity was assessed using the Australian standard (however, note that New Zealand streams do not have severe salinisation issues as in Australia, and it is therefore unlikely that New Zealand streams would approach this standard). Conductivity ranged from 198.4 to 298 µS/cm, well below the Australian upper limit of 1,500 µS/cm (Table 4.5) (Waterwatch Victoria, 1996). It should be noted that while conductivity was significantly higher at buffered areas, it was still well below the upper limit.

Turbidity ranged from 0.33 – 96.07 NTUs. Although these results have been skewed by the inclusion of high flow data, the ranges and the site means, whether the river was high or not, are well above the upper limits of 2 and 5.6 NTUs as set out by Davies-Colley and Wilcock (2004) and the ANZECC (2000) respectively (Table 4.6).

Dissolved oxygen ranged from 51.1 – 100.5 % saturation. The Third Schedule of the RMA (1991) recommends a level of 80 % saturation or above to ensure the protection of water in classes aquatic ecosystems (AE), fisheries (F), fish spawning (FS) and gathering or cultivation of shellfish for human consumption (SG) (Appendix 1). The ANZECC (2000) recommends dissolved oxygen saturation to fall between 98 % and 105 %, however this is considered to be overly stringent by some (eg. Milne and Perrie, 2006). The range of levels seen at the control and buffer sites at Boggy Creek, Harts Creek – The Lake Road and Harts Creek – Lochheads Road were all above 80 %. The means of the Birdlings Brook pair were below the 80 % level; however the mean at the planted site was just under and the upper range did overlap with the 80 % mark (Table 4.7). No sites ranges fell within the ANZECC (2000) limits however several sites did reach above 98 % (the lower range).

Phosphate levels ranged from 0.05 to 1.18 mg/L. These values are well above the levels of 0.01 and 0.03 mg/L as set out by the ANZECC (2000) and the Ministry for the Environment presented in Davies-Colley and Wilcock (2004) respectively (Table 4.8).

Nitrate levels ranged from 1 – 5.8 mg/L. These values are also well above the levels of 0.1 and 0.444 mg/L as set out by the as set out by Davies-Colley and Wilcock (2004) and the ANZECC (2000) respectively (Table 4.9).

Coliform levels ranged from 166.7 – 12466.7 CFUs/100 mL. All sites were well above the limit of 150 CFUs/100 mL set by the ANZECC (2000) (Table 4.10). *E. coli* levels ranged from 66.7 – 22733 CFUs/100 mL. The upper limit for *E. coli* concentration as set by the ANZECC (2000) is 126 CFUs/100 mL. While the lower ranges at the Birdlings Brook and Harts Creek – Lochheads Road pairs were within the 126 CFUs/100 mL, the means and upper ranges at all sites were far higher (Table 4.11). Although bacteria levels were affected by high flows, the ranges and the site means exceed guidelines whether the river was high or not.

The MCI values ranged from 57.1 to 127.1. Scores of 0 – 80 indicate probable severe pollution, 80 – 99 indicates probable moderate pollution, 100-119 indicates moderate enrichment and 120-200 indicates clean water (Stark & Maxted, 2007). The Birdlings Brook control site had MCI values that indicate probable severe pollution. The range of values at Harts Creek – The Lake Road planted indicate probable severe to moderate pollution. The Birdlings Brook planted site had MCI values that indicate probable moderate pollution. The Boggy Creek pair and the Harts Creek – The Lake Road control site all had values that span indicate probable moderate pollution to moderate enrichment. The Harts Creek – Lochheads Road pair had values that indicate moderate enrichment (Table 4.14).

No acceptable levels for salmonella, species richness or percentage EPT taxa could be found.

Overall, water quality was acceptable in terms of pH, temperature, conductivity and dissolved oxygen (excluding the Birdlings Brook pair). However, turbidity, dissolved reactive phosphate, nitrate-nitrogen and bacterial levels were unacceptable, and the MCI values indicated moderate enrichment to severe levels of pollution at all sites.

5.3 Effects of riparian restoration

Dissolved oxygen and conductivity were found to be significantly higher, and turbidity was significantly lower at sites where planting had occurred. While no other parameters measured showed overall significant differences, some were seen within site pairs between the control and buffer reaches.

The increase in dissolved oxygen and decrease in turbidity suggest that planted riparian buffers are having a positive impact on water quality in the Lake Ellesmere catchment. However, this positive impact is confounded by the significant increase in conductivity at planted sites. It is surprising that an improvement in turbidity was not associated with improvements in dissolved phosphorus and *E. coli*, as turbidities is usually an indicator of their presence (Davies-Colley & Smith, 2001).

These mixed responses to the planting of riparian buffers are not uncommon in New Zealand. The decrease in turbidity and lack of significant changes in nutrients and bacteria levels and macroinvertebrate communities observed in this study are consistent with those seen by Parkyn et al*.* (2003). However, this contrasts with the Jowett et al*.* (2009) study that showed reasonable improvements in macroinvertebrate communities over a 10-year period.

In this study, there are some plausible explanations for the mixed effects of riparian planting on water quality. Turbidity decreased in buffered reaches, while conductivity increased and nutrient concentrations did not change. Barling and Moore (1994) found that riparian buffer strips are more effective at removing coarse sediments than fine sediments from overland flow. The fine particles often remain suspended in surface runoff and can still enter the waterways (Barling & Moore, 1994). This may partly explain the observed differences between control and buffer reaches in this study.

Another potential explanation of the trends relates to the study design. In using a paired catchment design, it is important to consider that the water from the control reach upstream flows into the buffered reach. As a result, it is difficult to expect streams to recover in restored areas when nutrients and sediments are already in the flow from upstream sources (Scarsbrook & Halliday, 1998). This may explain the increase in conductivity at buffered sites as a result of accumulation between the control and buffer site at each pair. Based on the increased conductivity levels, it is likely that nutrients also continued to enter the waterway in the buffered area. However, the lack of increase may be attributed to in stream processing by macrophytes.

The trends in conductivity and nutrient levels show that gaps in the buffer system are still contributing to poor water quality. These gaps may be attributed to unplanted stream reaches or insufficient quality of planted buffers. Planting full stream reaches from the source at the headwaters down to the mouth would have the most significant effect in addressing this problem (Parkyn et al., 2003). Due to cost and the nature of private land ownership, this is often an unrealistic goal.

Discontinuous restoration is therefore a reality in catchments where land ownership does not currently allow for continuous buffers. It is successful in mitigating some effects, as exemplified by the reduction in turbidity in this study. However, gaps within buffers that contribute to poor water quality must be addressed. When assessing the effectiveness of restored riparian buffers, buffer width, fencing of control sites, timescales to restoration, and shading of the stream channel are the variables that must be considered.

5.3.1 Width of plantings

The widths of planted buffer areas ranged from 3m either side of the river at the Birdlings Brook site to 8.5m on one side of the river at Harts Creek – Lochheads Road (Table 3.1). Parkyn et al (2000) found that areas of planting less than 5m wide are not likely to support self-sustaining vegetation, and that weed control can be a problem in these situations. This can be seen at the Birdlings Brook and Boggy Creek planted sites (Figure 3.1 and Figure 3.4), where weed growth is occurring amidst the native planting.

A width of greater than 10m on either side of the waterway has been recommended as the minimum to meet terrestrial biodiversity restorative functions and suppress weed growth, and between 15 and 20m is likely to provide minimal maintenance and support self-sustaining vegetation (Davis & Meurk, 2001; Parkyn et al., 2000).

The insufficient buffer width seen at all sites is a likely contributing factor in the increased conductivity and unchanged nutrient and bacteria levels between control and buffer sites. Generally, the wider the planting the better the filtration ability (Fennessy and Cronk, 1997), which can be observed as increased sediment and nutrient removal (Parkyn, 2005).

5.3.2 Fencing to exclude stock

Fencing to exclude stock had occurred at the control reaches at Boggy Creek and Birdlings Brook, but at Harts Creek – The Lake Road and Lochheads Road fences were not in place. At Harts Creek – Lochheads Road the area adjacent to the stream was not actively grazed during the sampling period, while at Harts Creek – The Lake Road the paddock was actively grazed by cattle who were able to access the stream bed until a fence was constructed mid way through data collection in June 2010.

Research has shown that fencing can have immediate benefits to water quality and stream health. Stock access to stream banks results in trampling, that leads to bank erosion and deterioration and a subsequent loss of streamside habitat. In unfenced situations, stock also disturb the stream bed when they enter the waterway to drink, and deposit faecal matter on banks or directly in stream, affecting bacterial loadings (Waterwatch Victoria, 1996).

Given that all control sites are now fenced or not actively grazed by livestock, the immediate effects of stock exclusion may already be seen at the control sites as well as the buffered. The difficulty in finding non-fenced waterways that are actively grazed in Harts Creek, Boggy Creek and Birdlings Brook catchments is perhaps a sign that farmers are beginning to understand the benefits of riparian fencing. However, farmers may be more interested in the riparian planting's ability to prevent stock losses through drowning and a reduction in waterborne illnesses, rather than water quality improvements.

5.3.3 Stream shade

Establishing stream shade is likely to have an overall positive effect on water quality. It is reasonable to expect that temperature regimes would recover to cooler temperatures after flowing through closed canopy for a distance (Scarsbrook & Halliday, 1998). This is likely to have positive effects on dissolved oxygen levels and aquatic life (Parkyn et al., 2003).

A reduction in the amount of light reaching the streambed is also beneficial in reducing macrophyte growth. This is beneficial as it reduces the need for mechanical drainage clearance works which are commonplace in the Lake Ellesmere catchment. However, it has been suggested that this loss of instream vegetation could potentially result in an increase in nutrient levels, due to the loss of uptake by plants (Parkyn et al., 2003).

Full canopy cover had not been reached at any of the restored sites, although small reductions in the amount of light reaching the surface of the river were seen in most pairs (Table 4.1). A reduction in water temperature did not occur, and would not be expected until the buffers were more established, providing further shading to the river channel.

5.3.4 Influence of flow

Variation in stream flow has a significant influence on water quality (Hayward & Ward, 2008). High rainfall in the catchment results in high overland flow, which in turn increases the loading of nutrients and sediments within a waterway (Waterwatch Victoria, 1996).

It was found that flow rate massively influenced dependent variables, and therefore data collected on high flow dates was excluded from further analysis. In the future, more high flow data could be collected to allow further investigation of buffer effectiveness in high flow events. This was not possible with data from two sampling dates.

During high flow events, there appeared to be little difference in water quality between control and buffered sites. This is most likely due to the buffer having a small effect compared with the high amount of water entering the river as overland flow.

5.3.5 Long timescales to restoration

There is a large amount of uncertainty about how rapidly stream attributes will recover following restoration work. Jowett et al*.* (2009) found that it took several years for plantings to grow enough to have an impact on the stream habitat. They suggested that it could take in excess of 15 more years (in addition to the 10 year study period), before the reaches had reached states comparable to forest streams with overhead canopies and woody debris instream (Jowett et al., 2009).

Some aspects of water quality, particularly those that are sediment related, may improve rapidly following the initial exclusion of livestock (Davies-Colley et al., 2009). Other aspects such as nutrient levels may take much longer to recover. Shading and temperature adjusting can take from several years to decades, depending on stream channel width, and plant height and density (Davies-Colley et al., 2009). The significant decrease in turbidity and lack of improvements in nutrient and bacteria levels in tributaries to Lake Ellesmere is supported by the findings of Davies-Colley et al*.* (2009). Restoration of macroinvertebrate communities appears to be complex, and is dependent on a number of factors as discussed further below.

5.4 Invertebrate recolonisation

Community groups and other organisations initiating stream restoration projects usually focus on the improvement of physical habitat, such as riparian vegetation, bed substrate, pool-riffle sequences and flow variation (Blakely et al., 2006). In doing this, it is usually assumed that improving habitat is the key to biotic restoration (Bond & Lake, 2003). This has been coined the "field of dreams hypothesis", where if you create the habitat, the organisms will come (Palmer et al., 1997). However, Palmer et al*.* (1997) argue that in most cases, this is unlikely to be the situation. Ultimately, the success and time taken to achieve invertebrate community restoration is driven by the dispersal abilities of the organisms and the distance to the potential source populations (Smith, 2009). For this reason, it may be some time before species abundance and invertebrate community diversity is restored (Williams & Hynes, 1976).

MCI values in this study indicated moderate to severe levels of pollution, regardless of whether the site had been restored. There are a number of possible causes of this observation; a lack of source population, lack of dispersal or the habitat is not adequately restored.

There are four ways in which aquatic invertebrates act as source populations; downstream drift, upstream migration, migration from within substrate and oviposition via aerial dispersal (Blakely et al., 2006; Williams & Hynes, 1976). It is important that habitat restoration is placed at appropriate scales across space and time to facilitate this dispersal. Unfortunately, as is the case around Lake Ellesmere, degradation often occurrs over large areas, while restoration is focussed on one or a few sites (Bond & Lake, 2003). Even if source populations are present in the catchment, it is unlikely that they will travel large distances through unsuitable habitat to colonise restored areas. Therefore if invertebrate restoration is a primary goal, planted reaches should be located in targeted areas most likely to yield successful results. When designing habitat restoration projects these issues need to be considered more throughly than they often are (Bond & Lake, 2003; Winterbourn et al., 2007).

5.4.1 Impact of substrate on invertebrate habitat

In this study, mixed responses were seen in invertebrate community composition between control and buffer sites. These differences in composition appear to driven by the bed substrate present at the site, rather than whether the site is planted or not.

At sites where silt is present on the stream bed, it fills in the gaps between gravels and reduces habitat for invertebrates (Winterbourn et al., 2007). Sensitive species, including many EPT taxa are lost due to this loss of habitat. In addition, silt habitats support different macroinvertebrate communities made up of lower MCI scoring taxa compared with gravels (Stark, 1993). The combination of these factors result in lower index values.

It is likely that the responses seen in community composition in Figure 4.14, and the changes between the Birdlings Brook and Harts Creek – The Lake Road sites are affected by the presence of silt. These patterns can also be seen in Figure 4.15 and Figure 4.16 where the MCI and percentage EPT taxa is significantly lower at the Birdlings Brook control and Harts Creek buffer sites when compared within their pairs.

If invertebrate community restoration is a primary goal, the impact of substrate is something that should be considered when choosing sites for planting. Low gradient or discharge sites with silt present already are unlikely to be flushed through sufficiently to remove organic material and sediment. As a consequence, these areas will not be conducive to colonisation by some invertebrate taxa and perhaps restoration efforts should be focussed elsewhere.

5.5 Summary

This discussion has highlighted key findings and discussed the results of the study in the context of the literature. Riparian plantings were found to have a positive effect on water quality in terms of reducing turbidity and increasing dissolved oxygen, however nutrients, salts and bacteria were still able to move through restored areas and into waterways.

The effectiveness of riparian planting was evaluated in terms of the impact of impact of width, stream shade, flow, timescales and invertebrate recolonisation potential.

The background and findings of this study are summarised in the following conclusions chapter and suggestions for future research are made.

Chapter 6

Conclusions

Freshwater is a resource of global importance, yet its quality and availability is threatened. This can be seen in New Zealand, where around half of lowland water bodies fail to meet water quality standards. One of the main threats to water quality in New Zealand is the widespread conversion and intensification of land use. The Lake Ellesmere catchment is a typical example of these issues. A response that has emerged to combat water quality issues is the restoration of the riparian margin along waterways. Although riparian restoration is not a new phenomenon, little monitoring of its effectiveness has been undertaken. Without any measurement of success, it is difficult to evaluate the efficacy of riparian planting in protecting water quality.

This thesis aimed to establish whether riparian restoration was having a positive impact on chemical water quality and macroinvertebrate communities, using a case study of lowland streams in the Lake Ellesmere catchment. A paired catchment design was used to compare restored riparian buffers with unplanted control areas upstream, on four river reaches. In addition, baseline data was collected and field-sampling methods detailed, to allow future monitoring of these plantings as they continue to mature.

This study supports the findings of previous New Zealand case studies (Parkyn et al., 2003), suggesting that riparian restoration has a positive effect on water quality. This can be seen in the increase in dissolved oxygen and decrease in turbidity. However, the plantings that were studied fail to meet the recommended minimum width of 10 m, despite being praised as best examples in the catchment. This appears to have limited their effectiveness at protecting water quality, enabling the movement of nutrients, salts and bacteria through the buffer to the streams. A closed canopy had also not been achieved, therefore the expected result of decreased temperature and macrophyte die off was not seen.

Mixed responses were seen in invertebrate community composition, and it is likely that bed substrate, particularly the presence of silt, had a large effect on species present. Sites free from silt, or with little silt in the substrate are more likely to be recolonised by sensitive invertebrate species. This highlights the importance of considering the substrate in-stream when selecting sites for restoration, or alternatively the need for realistic goals to be set in terms of invertebrate community change.

It is clear that the effectiveness of riparian planting in protecting water quality is not as simple as whether the reach is planted or not. When planning restoration efforts and in the monitoring and evaluation of their effectiveness, a number of factors need to be considered. These factors include the length and width of buffer strip, time since retirement, stream shade, rainfall prior to sampling and sources of invertebrate colonisers.

Stream restoration is likely to be most effective when planting begins at the headwaters and is taken through to the river mouth. In situations where this is impractical, establishing buffers that are both long and wide is encouraged to protect tributaries.

While riparian restoration can mitigate some damage to aquatic systems caused by pastoral landuses, water quality problems may not be entirely solved. It is important for landowners and those undertaking restoration to remember that water quality reflects catchment landuse, and planted riparian buffers are a secondary prevention practice used in conjunction with managing field conservation practices. These include practices to reduce pollutant generation at the outset, compared with planted buffer strips that trap sediment and nutrients, which are a loss from the agricultural system.

Riparian management is a long-term task, and the beneficial results that it provides will take some time to become apparent. Some water quality parameters such as turbidity may recover quickly following fencing and planting. However, others such as nutrient levels may take much longer to improve. Because of this, there is a need for realistic goals to be set to avoid disappointment. It is likely that monitoring over a much longer timeframe is needed to fully assess changes in water quality as a result of restoration projects.

6.1 Scope for future research

This thesis has highlighted a number of areas that require further research.

Firstly, an exercise evaluating the tradeoffs between establishing a short but wide buffer compared with a long narrow buffer should be explored. Riparian restoration is a costly exercise, and by establishing the best ratio of buffer width and length to water quality outcomes, it would advise how funds be best spent for short and long term gain.

Furthermore, spatial analysis could be undertaken to evaluate the loss of productive land to planted area and the cost of this loss.

The effect of high flows on water quality in planted and unplanted areas warrants further research. In this study, water quality parameters varied little between planted and unplanted areas during high flow events. This raises the question whether plantings can be effective in protecting water quality in these situations.

A comprehensive study of the motives behind farmers fencing and planting waterways, and their perceived benefits of this could be undertaken. This type of study would be useful for understanding what motivates farmers to undertake riparian protection, and could be used in targeting areas for future planting.

Measurements of other indicators of restoration effectiveness, including fish monitoring, cultural health, public perceptions and social benefit would also be worthwhile.

6.2 Closing comments

This thesis has evaluated the efficacy of riparian plantings on water quality using a case study in the Lake Ellesmere catchment. Riparian restoration was found to have a positive effect on water quality in terms of increasing dissolved oxygen and decreasing turbidity. However, conductivity was found to increase at planted sites, and temperature, nutrient and bacteria levels showed no differences. Varied responses were seen in invertebrate communities, and it is likely that bed substrate had a considerable effect on the species composition between sites.

This research has shown the need for the systematic planning of riparian restoration efforts, and the need for realistic goals to be set. Ongoing monitoring is identified as being important in measuring the achievement of these goals, which can then provide feedback into further restoration efforts.

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Appendices

NOTE: The raw data presented in Appendices 5 & 6 is also contained in electronic format on the CD accompanying this thesis.

Appendix 1: Third Schedule of the Resource Management Act (1991) Water quality classes.

1 Class AE Water (being water managed for aquatic ecosystem purposes)

(1) The natural temperature of the water shall not be changed by more than 3° Celsius.

(2) The following shall not be allowed if they have an adverse effect on aquatic life:

- (a) Any pH change:
- (b) Any increase in the deposition of matter on the bed of the water body or coastal water:
- (c) Any discharge of a contaminant into the water.

(3) The concentration of dissolved oxygen shall exceed 80% of saturation concentration.

(4) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.

2 Class F Water (being water managed for fishery purposes)

(1) The natural temperature of the water—

- (a) Shall not be changed by more than 3° Celsius; and
- (b) Shall not exceed 25° Celsius.

(2) The concentration of dissolved oxygen shall exceed 80% of saturation concentration.

(3) Fish shall not be rendered unsuitable for human consumption by the presence of contaminants.

3 Class FS Water (being water managed for fish spawning purposes)

(1) The natural temperature of the water shall not be changed by more than 3° Celsius. The temperature of the water shall not adversely affect the spawning of the specified fish species during the spawning season.

(2) The concentration of dissolved oxygen shall exceed 80% of saturation concentration.

(3) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.

4 Class SG Water (being water managed for the gathering or cultivating of shellfish for human consumption)

(1) The natural temperature of the water shall not be changed by more than 3° Celsius.

(2) The concentration of dissolved oxygen shall exceed 80% of saturation concentration.

(3) Aquatic organisms shall not be rendered unsuitable for human consumption by the presence of contaminants.

5 Class CR Water (being water managed for contact recreation purposes)

(1) The visual clarity of the water shall not be so low as to be unsuitable for bathing.

(2) The water shall not be rendered unsuitable for bathing by the presence of contaminants.

(3) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.

6 Class WS Water (being water managed for water supply purposes)

(1) The pH of surface waters shall be within the range 6.0-9.0 units.

(2) The concentration of dissolved oxygen in surface waters shall exceed 5 grams per cubic metre.

(3) The water shall not be rendered unsuitable for treatment (equivalent to coagulation, filtration, and disinfection) for human consumption by the presence of contaminants.

(4) The water shall not be tainted or contaminated so as to make it unpalatable or unsuitable for consumption by humans after treatment (equivalent to coagulation, filtration, and disinfection), or unsuitable for irrigation.

(5) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.

7 Class I Water (being water managed for irrigation purposes)

(1) The water shall not be tainted or contaminated so as to make it unsuitable for the irrigation of crops growing or likely to be grown in the area to be irrigated.

(2) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.

8 Class IA Water (being water managed for industrial abstraction)

(1) The quality of the water shall not be altered in those characteristics which have a direct bearing upon its suitability for the specified industrial abstraction.

(2) There shall be no undesirable biological growths as a result of any discharge of a contaminant into the water.

9 Class NS Water (being water managed in its natural state)

The natural quality of the water shall not be altered.

10 Class A Water (being water managed for aesthetic purposes)

The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified aesthetic values.

11 Class C Water (being water managed for cultural purposes)

The quality of the water shall not be altered in those characteristics which have a direct bearing upon the specified cultural or spiritual values.

Appendix 2: Taxon scores used in calculating the MCI (Stark & Maxted, 2007).

INSECTA	
Ephemeroptera	
Acanthophlebia	7
Ameletopsis	10
Arachnocolus	8
Atalophlebioides	9
Austroclima	9
Austronella	7
Coloburiscus	9
Deleatidium	8
Ichthybotus	8
Isothraulus	8
Mauiulus	5
Neozephlebia	7
Nesameletus	9
Oniscigaster	10
Rallidens	9
Siphlaenigma	9
Tepakia	8
Zephlebia	7
Plecoptera	
Acroperla	5
Austroperla	9
Cristaperla	8
Halticoperla	8
Megaleptoperla	9
Nesoperla	5
Spaniocerca	8
Spaniocercoides	8
Stenoperla	10
Taraperla	7
Zelandobius	5
Zelandoperla	10
Megaloptera	
Archichauliodes	7
Odonata	
Aeshna	5
Antipodochlora	6
Austrolestes	6
Hemicordulia	6
Procordulia	6
Urupetala	5
Xanthocnemis	5
Hemiptera	
Anisops	5
Diaprepocoris	5
Microvelia	5
Sigara	5
Coleoptera	
Antiporus	5
Berosus	5
Copelatus	5
Dytiscidae	5
Elmidae	6
Enochrus	5
Hydraenidae	8
Hydrophilidae	5
Liodessus	5
Podaena	8
Ptilodactylidae	8
Rhantus	5
Scirtidae	8
Staphylinidae	5
Mecoptera	
Nannochorista	7
Neuroptera	
Kempynus	5

Diptera	
Anthomyiidae	3
Aphrophila	5
Austrosimulium	3
Calopsectra	4
Ceratopogonidae	3
Chironomidae	2
Chironomus	1
Cryptochironomus	3
Culex	3
Culicidae	3
Diptera indet.	3
Dixidae	4
Dolichopodidae	3
Empididae	3
Ephydridae	4
Eriopterini	9
Harrisius	6
Hexatomini	5
Limnophora	3
Limonia	6
Lobodiamesa	5
Maoridiamesa	3
Microchorista	4
Mischoderus	4
Molophilus	5
Muscidae	3
Neocurupira	7
Neolimnia	3
Nothodixa	4
Orthocladiinae	2
Parochlus	8
Paradixa	4
Paralimnophila	6
Paucispinigera	6
Pelecorhyncidae	9
Peritheates	7
Podonominae	8
Polypedilum	3
Psychodidae	1
Scatella	7
Sciomyzidae	3
Stratiomyidae	5
Syrphidae	1
Tabanidae	3
Tanypodinae	5
Tanytarsini	3
Tanytarsus	3
Thaumaleidae	9
Tipulidae	5
Zelandoptipula	6
Trichoptera	
Alloecentrella	9
Aoteapsyche	4
Beraeoptera	8
Confluens	5
Conuxia	8
Costachorema	7
Cryptobiosella	9
Diplectrona	9
Ecnomina	8
Edpercivalia	9
Ecnominidae	8
Helicopsyche	10
Hudsonema	6
Hydrobiosella	9
Hydrobiosis	5
Hydrochorema	9

Kokiria	9
Neurochorema	6
Oecetis	6
Oeconesidae	9
Olinga	9
Orthopsyche	9
Oxyethira	2
Paroxyethira	2
Philorheithrus	8
Plectrocnemia	8
Polyplectropus	8
Psilochorema	8
Pycnocentrella	9
Pycnocentria	7
Pycnocentrodes	5
Rakiura	10
Synchorema	9
Tiphobiosis	6
Triplectides	5
Triplectidina	5
Zelandoptila	8
Zelolessica	10
Lepidoptera	
Hygraula	4
Collembola	6
ACARINA	5
ARACHNIDA	
Dolomedes	5
CRUSTACEA	
Amphipoda	5
Cladocera	5
Copepoda	5
Isopoda	5
Ostracoda	3
Paracalliope	5
Paraleptamphopus	5
Paranephrops	5
Paratya	5
Tanaidacea	4
MOLLUSCA	
Ferrissia	3
Gyraulus	3
Hyridella	3
Latia	3
Lymnaeidae	3
Melanopsis	3
Physa	3
Physastra	5
Potamopyrgus	4
Sphaeriidae	3
OLIGOCHAETA	1
HIRUDINEA	3
PLATYHELMINTHES	3
NEMATODA	3
NEMATOMORPHA	3
NEMERTEA	3
COELENTERATA	
Hydra	3

Appendix 3: Grid references of sampling sites.

Site	Grid reference (Projection: NZTM 2000)
Boggy Creek Buffer	E1547764, N5154905
Boggy Creek Control	E1546037, N5156203
Harts Creek – The Lake Road Buffer	E1547401, N5150672
Harts Creek – The Lake Road Control	E1547176, N5150329
Harts Creek – Lochheads Road Buffer	E1543830, N5149919
Harts Creek – Lochheads Road Control	E1543520, N5150067
Birdlings Brook Buffer	E1543536, N5152616
Birdlings Brook Control	E1542371, N5153620

Appendix 4: Aerial photos of sampling sites.

A4.1: Study area. Aerial photo © 2009 Google, © 2010 Whereis® Sensis Pty Ltd, Image © 2011 GeoEye.

A4.2: Boggy Creek pair. Aerial photo © 2009 Google, © 2010 Whereis® Sensis Pty Ltd, Image © 2011 GeoEye.

A4.3: Harts Creek – The Lake Road pair. Aerial photo © 2009 Google, © 2010 Whereis® Sensis Pty Ltd, Image © 2011 GeoEye.

A4.4: Harts Creek – Lochheads Road pair. Aerial photo © 2009 Google, © 2010 Whereis® Sensis Pty Ltd, Image © 2011 GeoEye.

A4.5: Birdlings Brook pair. Aerial photo © 2009 Google, © 2010 Whereis® Sensis Pty Ltd, Image © 2011 GeoEye.

Appendix 5: Physical, chemical and microbiological raw data

NOTE: This data is also contained in electronic format on the CD accompanying this thesis.

A5.1: Boggy Creek buffer.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.3	7.2	7.1	7	6.7	6.8	6.7	6.9	7.3	7.4
Temperature °C	11.7	13	13.3	9.9	10.8	8.7	9	5.1	8.1	5.7
Conductivity µS/cm	293	290	308	303	317	387	409	376	377	246
Clarity NTUs	2.32	0.84	2.89	35.78	3.08	47.02	6.64	6.59	5.64	96.07
Dissolved oxygen % sat	92.4	91.4	91.3	95	92.3	84.8	90.1	96.9	97.6	85.2
Soluble phosphates mg/L	0.26	0.4	0.31	0.49	0.79	1.18	0.54	0.21	0.15	1.02
Soluble nitrates mg/L	3.2	3.3	3.5	3.8	3.5	2	3.7	3.5	3.4	1.2
Coliforms CFUs/100ml	1573	1100	583	1033	783	3233	2083	1833	1617	8600
E.coli CFUs/100ml	427	240	633	733	817	8300	317	433	583	22733
Salmonella CFUs/100ml	18	20	33	50	67	267	67	150	267	1200

A5.2: Boggy Creek control.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.3	7.4	7.2	7	6.4	6.8	6.9	6.8	7.2	7.4
Temperature °C	10.6	13	13.2	9.9	10.8	8.8	9.4	5.7	8.1	6
Conductivity µS/cm	295	290	307	305	311	401	399	367	369	262
Clarity NTUs	5.81	1.36	1.48	38.04	4.41	49.19	9.51	7.22	7.59	67.53
Dissolved oxygen % sat	87.9	85.4	80.3	86.8	87.2	87.2	92.5	97.8	100	85.7
Soluble phosphates mg/L	0.29	0.31	0.26	0.2	0.32	1.16	0.4	0.2	0.17	0.82
Soluble nitrates mg/L	2.4	3	3.3	4.1	3.1	3.2	3.2	2.9	1	1.2
Coliforms CFUs/100ml	582	1560	367	483	1100	3667	2700	967	1800	7600
E.coli CFUs/100ml	2564	840	2467	2550	5417	4750	867	1650	250	14467
Salmonella CFUs/100ml	45	0	0	50	100	150	150	83	133	733

A5.3: Harts Creek – The Lake Road buffer.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.1	7.2	7.2	7.1	6.6	6.8	6.9	7	7.1	7.1
Temperature ^{o}C	12.2	12.8	12.8	11.9	11.8	10.7	10.8	9.3	10.5	8.2
Conductivity µS/cm	200.8	198.4	203	204.8	209	256	276	263	275	271
Clarity NTUs	3.5	1.82	2.9	38.4	3.89	17.8 2	8.24	6.43	4.95	72.04
Dissolved oxygen % sat	100.5	98.3	89.5	91.2	86.3	85.1	85.7	91.4	91.1	82.2
Soluble phosphates mg/L	0.21	0.23	0.21	0.35	0.25	0.18	0.62	0.26	0.14	0.66
Soluble nitrates mg/L	3	3.3	3	3.1	2.2	3.8	3.1	3.4	4.9	2.8
Coliforms CFUs/100ml	582	680	283	3833	800	3100	1667	683	617	1246 7
E.coli CFUs/100ml	455	460	233	917	600	1233	400	333	150	1066 7
Salmonella CFUs/100ml	45	20	67	17	67	67	133	33	833	1333

A5.4: Harts Creek – The Lake Road control.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.4	7.2	7.2	7.1	6.5	6.8	6.9	7.1	7.4	7.2
Temperature ^{o}C	11.9	12.3	12.6	11.8	12	10.8	10.9	9.5	10.5	8.2
Conductivity µS/cm	200.5	201.6	204.1	205.7	209.5	255	275	263	276	273
Clarity NTUs	12.31	3.96	2.12	40.58	4.16	17.86	5.45	6.41	6.43	73.36
Dissolved oxygen % sat	96	92.6	87.6	89	88.7	84.2	84.7	89.3	89.1	81.4
Soluble phosphates mg/L	0.2	0.34	0.1	0.23	0.19	0.1	0.21	0.1	0.16	0.48
Soluble nitrates mg/L	2.2	3.3	3.1	2.8	2.3	3.6	2.9	3.2	2.2	2.4
Coliforms CFUs/100ml	464	340	483	983	233	3467	1567	700	533	12267
E.coli CFUs/100ml	373	620	383	1450	733	1717	433	250	183	9933
Salmonella CFUs/100ml	18	20	33	17	33	33	233	17	50	1000

A5.5: Harts Creek – Lochheads Road buffer.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.4	7	7	6.9	6.8	6.8	6.8	7	7.4	7.2
Temperature °C	12.8	13.4	12.8	12.6	12.3	11	11.1	10.4	11.3	9.7
Conductivity µS/cm	244	244	246	244	246	250	250	248	249	240
Clarity NTUs	3.38	3.67	1.92	42.6	2.42	10.89	3.72	0.84	1.9	22.72
Dissolved oxygen % sat	91.3	88	83.9	87.2	87.1	85.2	86.2	90.6	91.9	88.4
Soluble phosphates mg/L	0.08	0.21	0.23	0.19	0.37	0.27	0.25	0.21	0.1	0.38
Soluble nitrates mg/L	3.8	3	4.5	5	3.3	3.8	3.5	4.5	3.2	4
Coliforms CFUs/100ml	382	460	450	583	317	2883	1100	300	2417	8200
E.coli CFUs/100ml	282	100	167	117	233	5367	500	333	167	2000
Salmonella CFUs/100ml	18	20	0	67	0	200	183	33	183	1133

A5.6: Harts Creek – Lochheads Road control.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.4	7.2	7.1	6.6	6.6	6.7	7	6.9	7.3	7.2
Temperature °C	13	13.4	12.9	12.4	12.3	11	11.1	10.5	11.3	9.8
Conductivity µS/cm	243	243	246	245	246	251	250	247	249	239
Clarity NTUs	4.54	0.97	1.65	44.41	5.63	12.61	4.05	0.69	1.06	19.39
Dissolved oxygen % sat	89	85.6	82.9	85.5	86.8	84.4	85.1	89.4	91.1	87.9
Soluble phosphates mg/L	0.05	0.15	0.35	0.22	0.18	0.22	0.22	0.16	0.2	0.22
Soluble nitrates mg/L	3.2	2.8	4.7	4.5	4.1	4	4.2	4.2	4.1	4.6
Coliforms CFUs/100ml	491	220	250	317	167	3750	2100	417	3250	8000
E.coli CFUs/100ml	264	140	250	183	117	5283	500	183	100	1133
Salmonella CFUs/100ml	64	0	17	33	0	150	167	17	50	800

A5.7: Birdlings Brook buffer.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7	6.9	7.2	6.8	6.6	6.6	7	6.9	7	7.2
Temperature °C	13.2	13.3	13.3	11.7	11.9	10.2	10.3	8.4	9.9	8.5
Conductivity µS/cm	320	318	321	313	311	391	400	376	385	336
Clarity NTUs	4.58	0.33	1.62	46.15	3.06	29.72	6.53	6.29	7.92	73.87
Dissolved oxygen % sat	93.6	78.5	71.2	76.6	75.7	75.4	79.8	84.8	86.3	70.4
Soluble phosphates mg/L	0.13	0.56	0.33	0.17	0.28	0.38	0.36	0.8	0.14	0.36
Soluble nitrates mg/L	2.6	2	4.5	2.7	3.4	4	4.6	4.4	5.8	2.8
Coliforms CFUs/100ml	2227	800	483	867	367	4267	1467	567	500	9333
E.coli CFUs/100ml	173	200	150	483	67	1850	550	433	133	4067
Salmonella CFUs/100ml	36	20	33	33	50	67	83	50	0	867

A5.8: Birdlings Brook control.

	30/03	13/04	27/04	11/05	25/05	8/06	22/06	13/07	27/07	10/08
pH	7.2	7.1	7	6.9	6.5	6.3	6.6	6.6	7.1	7.1
Temperature °C	15.7	13.4	13.1	12.3	12.3	10.5	10.4	8.7	10.2	8.2
Conductivity µS/cm	295	299	299	292	296	402	414	370	387	344
Clarity NTUs	12.74	2.67	19.88	52.06	4.69	91.81	9.44	3.16	6.15	43.52
Dissolved oxygen % sat	74.4	51.1	65.1	75.7	69	53.6	66.3	70.8	75.9	76.1
Soluble phosphates mg/L	0.1	0.3	0.66	0.25	0.76	0.51	0.23	0.72	0.34	0.36
Soluble nitrates mg/L	2.1	3.6	3	3.9	3.4	3.2	4.2	4	2.2	3.4
Coliforms CFUs/100ml	582	2000	3350	1167	300	5667	3167	867	683	8267
E.coli CFUs/100ml	582	200	67	100	217	6067	500	67	83	1600
Salmonella CFUs/100ml	18	0	33	183	17	117	100	67	0	1200

Appendix 6: Macroinvertebrate raw data.

NOTE: This data is also contained in electronic format on the CD accompanying this thesis.

This data is presence/absence data, 1 indicates that the species was found at the site on the sampling date

A6.1: Boggy Creek buffer.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus					
Deleatidium	1	1	1	1	1
Aoteapsyche			1		1
Helicopsyche					
Hudsonema	1	1	1	1	1
Hydrobiosis					
Neurochorema					
Oeconesidae					
Olinga					
Oxyethira					
Polyplectropus					
Psilochorema	1	1		1	1
Pycnocentria					
Pycnocentrodes	1	1	1	1	1
Triplectides	1			1	1
Zelandobius					
Austrolestes					
Nannochorista					
Elmidae	1	1	1	1	1
Microvelia		1			
Sigara					
Austrosimulium					
Chironomus					
Hexatomini					1
Mischoderus					
Muscidae					
Orthocladiinae					
Sciomyzidae					
Stratiomyidae					
Gyraulus					
Physa	1	1	1		
Potamopyrgus	1	1		1	1
Sphaeriidae				1	
Amphipoda					
Isopoda					
Ostracoda	1	1	1	1	1
Hirudinea	1				
Oligochaeta	1	1			1
Platyhelminthes	1				1
Species richness	12	10	7	9	12
MCI	91.67	90	100	113. 33	96.67
% EPT taxa	41.67	40	57.14	55.56	50

A6.2: Boggy Creek control.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus					
Deleatidium	1	1	1	1	1
Aoteapsyche	1			1	1
Helicopsyche	1				
Hudsonema				1	
Hydrobiosis				1	
Neurochorema					
Oeconesidae					
Olinga					
Oxyethira					
Polyplectropus					
Psilochorema	1		1		
Pycnocentria					
Pycnocentrodes	1	1	1	1	1
Triplectides	1	1	1		
Zelandobius					
Austrolestes					
Nannochorista					
Elmidae	1	1	1	1	1
Microvelia					
Sigara					1
Austrosimulium		1	1		
Chironomus					
Hexatomini					
Mischoderus					
Muscidae					
Orthocladiinae					
Sciomyzidae					
Stratiomyidae					
Gyraulus			1		
Physa				1	
Potamopyrgus			1	1	1
Sphaeriidae				1	
Amphipoda	1	1	1	1	
Isopoda					
Ostracoda		1	1	1	
Hirudinea					
Oligochaeta	1		1	1	1
Platyhelminthes		1	1		1
Species richness	9	8	12	12	8
MCI	115.56	95	90	90	90
% EPT taxa	66.67	37.5	33.33	41.67	37.5

A6.3: Harts Creek – The Lake Road buffer.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus					
Deleatidium					
Aoteapsyche					
Helicopsyche	1				
Hudsonema					
Hydrobiosis					
Neurochorema					
Oeconesidae					
Olinga					
Oxyethira	1	1			
Polyplectropus					
Psilochorema					
Pycnocentria	1		1	1	
Pycnocentrodes		1			1
Triplectides		1	1	1	1
Zelandobius					1
Austrolestes		1	1	1	1
Nannochorista					
Elmidae	1	1			
Microvelia					
Sigara		1	1		
Austrosimulium					
Chironomus		1			1
Hexatomini					
Mischoderus					
Muscidae					
Orthocladiinae					
Sciomyzidae					
Stratiomyidae					
Gyraulus	1	1	1	1	
Physa	1	1	1	1	1
Potamopyrgus	1	1	1	1	1
Sphaeriidae	1				1
Amphipoda	1	1	1	1	1
Isopoda					
Ostracoda	1	1	1	1	1
Hirudinea	1		1		
Oligochaeta				1	1
Platyhelminthes			1		
Species richness	11	12	11	9	11
MCI	89.09	78.33	85.45	88.89	74.55
% EPT taxa	27.27	25	18.18	22.22	27.27

A6.4: Harts Creek – The Lake Road control.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus	1		1	1	1
Deleatidium	1	1	1	1	1
Aoteapsyche			1		1
Helicopsyche			1		1
Hudsonema					
Hydrobiosis		1			1
Neurochorema				1	
Oeconesidae					
Olinga	1		1	1	1
Oxyethira					
Polyplectropus					
Psilochorema	1		1		
Pycnocentria	1	1	1	1	1
Pycnocentrodes	1	1	1	1	1
Triplectides					1
Zelandobius					1
Austrolestes					
Nannochorista					
Elmidae	1		1	1	
Microvelia					
Sigara					
Austrosimulium					
Chironomus		1			
Hexatomini					
Mischoderus					
Muscidae				1	
Orthocladiinae					
Sciomyzidae					
Stratiomyidae					
Gyraulus					
Physa	1	1	1	1	
Potamopyrgus	1	1	1	1	1
Sphaeriidae	1	1			
Amphipoda	1	1		1	1
Isopoda		1			
Ostracoda	1	1		1	
Hirudinea					
Oligochaeta	1	1	1	1	1
Platyhelminthes	1		1	1	1
Species richness	14	12	13	14	14
MCI	105.71	85	118.46	102.86	115.71
% EPT taxa	42.86	33.33	61.54	42.86	71.43

A6.5: Harts Creek – Lochheads Road buffer.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus	1	1	1	1	1
Deleatidium	1	1	1	1	1
Aoteapsyche		1	1	1	1
Helicopsyche	1	1	1	1	1
Hudsonema					
Hydrobiosis	1	1	1	1	
Neurochorema					
Oeconesidae					
Olinga	1	1	1	1	1
Oxyethira					
Polyplectropus	1				
Psilochorema	1	1			1
Pycnocentria	1	1	1	1	1
Pycnocentrodes	1	1	1	1	1
Triplectides					
Zelandobius					1
Austrolestes					
Nannochorista					
Elmidae	1		1		
Microvelia					
Sigara					
Austrosimulium				1	
Chironomus		1			
Hexatomini					
Mischoderus					
Muscidae					
Orthocladiinae					
Sciomyzidae					
Stratiomyidae					
Gyraulus					
Physa					
Potamopyrgus	1	1	1	1	1
Sphaeriidae	1				
Amphipoda	1	1	1	1	1
Isopoda			1		
Ostracoda			1		
Hirudinea					
Oligochaeta	1	1		1	1
Platyhelminthes				1	1
Species richness	14	13	13	13	13
MCI	127.14	118.46	124.62	113.85	120
% EPT taxa	64.29	69.23	61.54	61.54	69.23

A6.6: Harts Creek – Lochheads Road control.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus		1			
Deleatidium	1	1	1	1	1
Aoteapsyche					1
Helicopsyche		1	1	1	
Hudsonema					
Hydrobiosis		1			
Neurochorema			1		
Oeconesidae					1
Olinga	1	1	1	1	
Oxyethira					
Polyplectropus					1
Psilochorema	1	1	1	1	1
Pycnocentria		1	1	1	1
Pycnocentrodes	1	1	1	1	
Triplectides	1		1		1
Zelandobius					
Austrolestes					
Nannochorista			1		1
Elmidae					1
Microvelia					
Sigara					
Austrosimulium				1	
Chironomus					
Hexatomini				1	
Mischoderus	1				
Muscidae					
Orthocladiinae					
Sciomyzidae					
Stratiomyidae					
Gyraulus					
Physa					
Potamopyrgus			1		
Sphaeriidae					
Amphipoda		1	1	1	1
Isopoda					
Ostracoda		1	1		1
Hirudinea					
Oligochaeta	1	1	1	1	1
Platyhelminthes	1	1	1	1	1
Species richness	8	12	14	11	13
MCI	107.5	123.33	115.71	116.36	113.85
% EPT taxa	62.5	66.67	57.14	54.55	53.85

A6.7: Birdlings Brook buffer.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus					
Deleatidium	1	1	1	1	1
Aoteapsyche					
Helicopsyche					
Hudsonema	1	1	1		
Hydrobiosis			1	1	
Neurochorema					
Oeconesidae					
Olinga					
Oxyethira					
Polyplectropus	1	1			
Psilochorema				1	
Pycnocentria	1			1	
Pycnocentrodes	1	1	1		
Triplectides	1	1	1		
Zelandobius					
Austrolestes		1	1		1
Nannochorista					
Elmidae				1	
Microvelia		1			
Sigara					
Austrosimulium					
Chironomus					
Hexatomini					
Mischoderus					
Muscidae					
Orthocladiinae				1	
Sciomyzidae					
Stratiomyidae					
Gyraulus	1	1		1	
Physa	1	1	1	1	1
Potamopyrgus	1	1	1	1	1
Sphaeriidae	1				
Amphipoda	1	1	1	1	1
Isopoda					
Ostracoda		1	1	1	1
Hirudinea					
Oligochaeta	1	1	1	1	1
Platyhelminthes				1	
Species richness	12	13	11	13	7
MCI	96.67	93.85	94.55	90.77	85.71
% EPT taxa	50	38.46	45.45	30.77	14.29

A6.8: Birdlings Brook control.

	29/03	26/04	24/05	21/06	26/07
Coloburiscus					
Deleatidium					
Aoteapsyche					
Helicopsyche					
Hudsonema					
Hydrobiosis					
Neurochorema					
Oeconesidae					
Olinga					
Oxyethira					
Polyplectropus					
Psilochorema					
Pycnocentria					
Pycnocentrodes		1			
Triplectides		1	1		
Zelandobius					
Austrolestes					
Nannochorista					
Elmidae		1			
Microvelia	1	1			
Sigara	1	1	1		
Austrosimulium					
Chironomus	1	1	1		1
Hexatomini				1	1
Mischoderus					
Muscidae					
Orthocladiinae				1	
Sciomyzidae					1
Stratiomyidae			1		
Gyraulus	1				
Physa		1	1	1	
Potamopyrgus	1	1	1	1	1
Sphaeriidae	1	1	1		1
Amphipoda				1	
Isopoda					
Ostracoda		1	1		1
Hirudinea					
Oligochaeta		1	1	1	1
Platyhelminthes		1			
Species richness	6	12	9	6	7
MCI	70	73.33	66.67	66.67	57.14
% EPT taxa	0	16.67	11.11	0	0

Appendix 7: Flow data.

A7.1: Flow classes.

Mean flow over sampling period (l/s)	1275.2
Standard deviation	632.3
Low flow	Q ≤ 642.9
Medium flow	642.9 < Q < 1907.5
High flow	Q ≥ 1907.5

A7.2: Flow characterisations.

Date	Flow at Harts Creek Timberyard Road (l/s)	Flow characterisation
24/3	865	Medium
13/4	890	Medium
27/4	939	Medium
11/5	957	Medium
25/5	1036	Medium
8/6	4125	High
22/6	1618	Medium
13/7	1561	Medium
27/7	1783	Medium
10/8	2529	High

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Information supplied to: **Katie Collins**
Description of information to which these terms and conditions apply:
Daily Mean flow data for the following sites:

- 68322: Harts Creek at Timber Yard Road (Grid Reference: M36:571-119) from the 1st of February 2010 to the 12th of August 2010

All data for this site is provisional because data has not been audited. Data from the 8th of June to 12th of August is provisional as it has not been quality checked by hydro staff.

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Prepared by: Kerrie Osten	Date: 13th August 2010