THE EFFECTS OF WILLOW AND WILLOW CONTROL ON WETLAND MICROFAUNAL ASSEMBLAGES IN SOUTH TAUPO WETLAND

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ABSTRACT

Few studies have examined microfaunal assemblages living among *Salix cinerea* infestations in freshwater wetlands, or their responses to willow control treatment. The aim of this research was to quantitatively examine microfaunal assemblage abundance, richness and community composition among *S. cinerea* stands within the South Taupo Wetland, and determine whether these microfaunal assemblages are affected by willow growth and willow control treatment.

Long-term effects of microfaunal community composition between native vegetation versus live and dead *S. cinerea* were examined in two blocks of the Waiotaka Scenic Reserve. Microfauna and physicochemical sampling were performed on three occasions to assess any seasonal effects on community composition. Results indicated there were no significant differences of physicochemical variables amongst natives, live and dead *S. cinerea*, with the exception of dissolved oxygen in late summer and canopy density in all seasons. This could be due to the *S. cinerea* trees representing stand-alone individuals, with a continuous canopy not yet formed. Overall, apart from shading and dissolved oxygen levels, environmental conditions of *S. cinerea* stands in this study seemingly made no significant difference to environmental variables.

The abundant taxa found in the study were copepods, cladocerans and ostracods along with diverse species of rotifers, including the first record for New Zealand of the rotifer species *Tetrasiphon hydracora*. ANOVA indicated that there were no significant differences in microfaunal species richness between native, live and dead *S. cinerea* in any season. However, MDS ordination and ANOSIM results of species composition indicated that microfaunal assemblages were clustered in groups either side of the sand bar, suggesting that Blocks 1 and 2 functioned independently. This may be influenced by hydrological differences between Block 1 and 2 of the wetland reserve, with differing responses to fluctuating lake levels and seasonal rainfall, suggesting that microfaunal communities are regulated by hydrology rather than by the presence of willows or willow control.

Short-term effects of microfaunal community composition post willow control treatment were examined in Block 2 of the Waiotaka Scenice Reserve. Microfauna and physicochemical sampling were performed before and after treatment to assess effects on community composition post willow control treatment, using ground control method of drill and inject with a herbicide mix of metsulfuron. No significant differences in environmental variables were observed post treatment, with the exception of canopy density cover. Treated *S. cinerea* trees died and lost their leaves after ground application of metsulfuron. Microfaunal abundance and diversity were low before and treatment, suggesting that the application of metsulfuron made little difference to microfaunal assemblages.

Overall, the findings of this study indicate that the presence of *S. cinerea* seemed to make no significant difference to microfaunal abundance and diversity, possibly due to stand alone individuals rather than the formation of a dense canopy. Furthermore, ground control treatment of *S. cinerea* using metsulfuron had no direct or indirect impacts to microfaunal abundance and diversity. However, had the study been undertaken under a dense canopy of *S. cinerea* it is likely that the results may potentially be different.

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Māmā loves you my pēpi, you are my inspiration xox

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CHAPTER ONE: INTRODUCTION

1.1 Wetlands

The term wetland integrates a wide range of inland, coastal and marine habitats, where the flora and fauna that inhabit these wetlands are uniquely adapted to tolerate variable environmental conditions (Bacon 1997). The Ramsar Convention 1971, an intergovernmental treaty, broadly defines wetlands as *"areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres*" (Hails 1997).

Wetlands are amongst the most complex ecosystems in the world, supporting habitats for distinctive flora and fauna, both aquatic and terrestrial (Gren et al. 1994; Hails 1997), particularly those adapted to transitional areas between aquatic and terrestrial ecosystems (Mitsch & Gosselink 2007). They play a vital role for diverse species of plants, invertebrates, amphibians, reptiles and mammals that depend on wetland ecosystems for food, habitat and shelter (Mitra et al. 2005; Mitsch & Gosselink 2007).

Wetlands are important globally in nutrient cycling, waste filtration, sediment accretion and erosion control (Mitsch & Gosselink 2007). They act as sinks for inorganic nutrients and as sources of organic materials for downstream or adjacent ecosystems. They have the capacity to improve water quality, by filtering wastes and reducing the transport of organic material, sediments and toxic substances to adjacent water bodies (Gren et al. 1994). They can stabilise water supply, ameliorating both floods and droughts, protect shorelines and recharge aquifers (Mitsch & Gosselink 2007).

Wetlands have experienced intensive modifications by human impacts, with only a small percentage of wetlands remaining globally following intensive development and urbanisation (Qin & Mitsch 2009). In Western Society, the drainage and destruction of wetlands became an accepted practice throughout the world and has even been encouraged by specific government policies (Mitsch

2002). The loss and degradation of wetlands include the damming of rivers, disconnection of floodplain wetlands from flood flows, eutrophication, contamination, the practices of filling, dyking and draining, and the invasion of exotic plant species (Brinson & Malvarez 2002). The loss of wetlands has continued despite the countless values that they provide both ecologically and economically. Mitsch and Gosselink (2000) illustrated that 50 percent of the original wetlands worldwide had been destroyed. High rates of wetland loss have been recorded due to European settlement in the lower United States, Europe, Australia, Canada and China (Mitsch & Gosselink 2000).

Wetlands have been destroyed at alarming rates throughout the world. Fortunately, there has been a shift in worldwide recognition of the fundamental role and values of wetlands, resulting in legislation amendments in many parts of the world (Mitsch & Gosselink 2007). The Convention on Wetlands of International Importance was adopted at Ramsar, a city on the Iranian shores of the Caspian in 1971. The treaty has influenced worldwide action at the governmental level for the conservation and wise use of wetlands (Hails 1997). Ramsar established, for the first time, an international convention, with two fundamental ideals: a list of wetlands of international importance and the principle of wise use of all wetlands in the territory of a Contracting Party (Smart 1997). The Convention became an effective instrument to ensure the conservation and wise use of wetlands worldwide.

1.1.1 Freshwater Wetlands in New Zealand

A definition of wetland for New Zealand purposes is provided in the Resource Management Act (1991); "*Wetlands" includes permanently and intermittently wet areas, shallow water, and land water margins that support a natural ecosystem of plants and animals that are adapted to wet conditions".*

New Zealand wetlands are among the most threatened ecosystem types in the country, with only an estimated 10% of the original wetland cover remaining (Cromarty & Scott 1995). Most of the original wetland areas have been converted for agriculture, while others have been adversely affected by hydroelectricity

generation and flood protection (Eser 1998) and have been extensively modified due to nutrient enrichment and the introduction of exotic weed species, especially in lowland areas (Singers 2009).

In 1976 New Zealand signed the Ramsar Convention and implemented the treaty by instantly designating two wetland sites for protection under the convention (Gerbeaux 2002). By 2006, New Zealand had established six sites designated under the Ramsar Convention as Wetlands of International Importance. These wetlands were selected due to their international significance of ecology, botany, zoology, limnology and hydrology, as well as meeting the specified criteria outlined in by the Ramsar Convention on Wetlands. The Department of Conservation (DOC) is the New Zealand agency which administers the Ramsar Convention sites (Department of Conservation 2010).

The major functional wetland types in New Zealand are bog, fen, swamp, marsh and shallow water freshwater wetlands (Johnson & Gerbeaux 2004; Peters & Clarkson 2010). These classes are controlled by distinctive combinations of substrate, water regimes, nutrient status, pH and peat content. These wetlands vary from the more fertile eutrophic swamps to oligotrophic peat bogs (Johnson & Gerbeaux 2004).

New Zealand wetlands are most at risk of weed invasion due to their low stature native vegetation communities and fragmented nature (Owen 1998) which has increased the likelihood of weed access and invasion. Invasive weeds can threaten the long term survival of native plant communities resulting in the displacement of native species, modification of successional processes and alter the structure and composition of native vegetation (Owen 1998). Species of the genus *Salix* have been ranked as the most invasive exotic weed threatening New Zealand wetlands (Champion et al. 2008).

1.2 Salix cinerea

Salix species originates from northern temperate regions of Europe, western Asia and northern Africa and has established in Southern Hemisphere countries including Australia and New Zealand (Harman 2004). *Salix* is one of four genera in the Family Salicaceae and is known to have between 300 and 500 species. Willows are usually divided into three subgenera: *Salix* (tree willow), *Caprisalix* (shrub willows – sallows and osiers), and *Chamaetia* (dwarf, arctic or alpine willows) (Harman 2004). There are two subspecies, *S. cinerea* supsp. *cinerea* distributed in Central and Eastern Europe, Western Asia, and the *S. cinerea* supsp. *oleifolia* (Smith) Macreight (syn. *S. atrocinerea* Brot) located in Western Europe and Northwest Africa (Christensen & Nielsen 1992). Most of these species have northern temperate distributions, although there are a few species in temperate zones of the Andes and upland central and southern Africa (Thompson & Reeves 1994). *Salix* can occupy a range of aquatic environments, which can be divided into two major groups; alluvial or riparian (along rivers and streambanks) and wetlands (open water and saturated soils) (Kuzovkina & Quigley 2005).

Salix cinerea, or grey willow, is a deciduous shrub or small tree known to reach up to 10 m high. It forms dense thickets and has sympodial growth, where the apical meristem often dies during winter and vegetative growth continues from the nearest lateral meristems (Alliende & Harper 1989). This increases branch angling as the individual matures with lower branches becoming horizontal and reduces self-shading. Branches at the base of the tree are flexible and branchlets are grey or greenish grey (Webb et al. 1988). The buds are reddish and glabrate. The petiole can grow up to 1 cm long on adult shoots but are usually short and hairy. Leaves can range from 2-7 cm x 1.5-3.5 cm, usually smaller at the base of lateral shoots, glandular and obovate to elliptic in shape (Webb et al. 1988). Flowering of *S. cinerea* occurs during early spring and matures in early summer, the catkins (flower clusters) appear before the leaves, on both sexes, and are broad-cylindric to cylindric-ovate in shape at 1.5-3.5 cm long. Flower bracts are 1.5-3.0 mm long, brown to black in colour with an obtuse to rounded apex (Webb et al. 1988).

Salix cinerea is typical of fen peats and lowland marshes, wet forests, alpine bogs and disturbed lands where flood and fire has occurred resulting in the opening of native plant canopy. It is tolerant of permanent waterlogging and associated anaerobic conditions, and is found over a wide range of soils from nutrient rich mineralised soils to acid peaty soils with a pH as low as 3.5 (Champion 1994).

Salix cinerea are fast growing trees due to their massive root system penetrating deep into the soil, perennial character, high evapotranspiration, rapid regeneration, simple vegetative reproduction and immense biomass production (Sottnikova et al. 2003). *Salix cinerea* can be found in waterlogged soils containing high levels of exchangeable manganese and iron, and has a capability of tolerating contaminated environments in waterlogged soils, due to its root system reducing the solubility of manganese and iron compounds (Talbot et al. 1987).

In the Northern Hemisphere, where *Salix* species are native, they provide various ecological benefits. Salix are versatile and ideal for river training and erosion control. Their wide spreading fibrous root systems help to bind soil on stream-sides and hillsides (Russell 1994). *Salix* species do not cause threats in their native countries as they are adapted to the confines of their native environments. In Europe and Japan riparian willows regenerate from seed and germinate in bare, wet sediments along the river banks, forming the meander over floodplains. In some of their native countries, willows are succeeded by other hardwoods and conifers that regenerate in the shade of willows, restricting their spread among other native plant communities (Cremer 2003).

In the Southern Hemisphere *Salix* species have become widespread, highly invasive, and many have caused substantial ecological and economic losses in wetland ecosystems (Adair et al. 2006). Due to geographical and evolutionary isolation, both New Zealand and Australia have evolved biota with a major element of endemism, making these countries susceptible to the invasion of exotic plants (Williams & West 2000). The introduction of *Salix* species to New Zealand and Australia was deliberate and have since become naturalised, forming self-sustaining populations and threatening indigenous plant communities (Williams & West 2000).

In Australia, species of *Salix* were introduced to the Southern areas of Australia for their ability to stabilise river banks for flood protection. They were also planted in areas of cleared land for farming purposes, where native species found it difficult to establish (Cremer 2003). Unfortunately, most species of *Salix* became widespread and highly invasive causing significant ecological and economic losses in wetland systems. Willows have been rated in the Weeds of National Significance and subjected to major control efforts to minimise their environmental impact (Adair et al. 2006). *Salix cinerea* has become a dominant feature in many reserves and wetlands throughout Victoria, Tasmania, New South Wales and South Australia (Ward et al. 2002). In Victoria, *S. cinerea* has spread throughout riparian habitats, brackish wetlands, wet forests and alpine bogs (Cremer 2003).

1.2.1 *Salix cinerea* in New Zealand

In New Zealand, species of *Salix* were introduced in the early nineteenth century (Thompson & Reeves 1994; van Kraayenoord C.W.S & Hathaway 1986). At least eleven species of *Salix* and five hybrids have been naturalised in New Zealand. Various species were planted along water ways for their ability to provide erosion protection of riverbanks and for soil conservation purposes. The earliest records of *S. babylonica* were planted on the Akaroa Peninsula, Christchurch, on land farmed by French settlers. By the 1860s it had been documented in Northland, planted along the Northern Wairoa River and known to have spread along the river by the 1880s. Another species, *S. fragilis*, was introduced by the 1860s and had naturalised by 1879. It was planted in the South Canterbury, Hawkes Bay and Waikato regions to promote river bank protection. By the 1870s *S. cinerea* were first introduced to the South Island, followed by the North Island, and was later reported in the 1940s in the Waikato (Thompson & Reeves 1994).

The dispersal of vegetative propagation and rapid growth of most *Salix* species have resulted in their widespread distribution (Webb et al. 1988). The persistence of *S. cinerea* is demonstrated in its effective dispersal mechanism. The seeds are small and adapted to long-distance wind dispersal due to the plume of hairs attached to its base. Although the seed has a high light requirement and are short

lived, they demonstrate high germination rates during major flooding, siltation and fire events (Champion 1994).

As seen in the Australian context, most of these *Salix* species established and became highly invasive, disturbing ecological succession of freshwater ecosystems throughout New Zealand. As with elsewhere, the successful invasion of *S. cinerea* throughout New Zealand wetlands is due to its ability to tolerate a variety of environmental conditions, effective seed dispersal mechanism and the displacement of indigenous plant communities (Webb et al. 1988). There are few wetlands in the country that have not been colonised by *S. cinerea* (de Winton & Champion 1993). High density *S. cinerea* invasions include wetlands in the Bay of Plenty, the eastern South Island and the Waikato (Webb et al. 1988). In the New Zealand wetland setting, *S. cinerea* is typically found in areas of open water communities, among raupo (*Typha orientalis*), *Carex secta*, rushes, sedges, small wetland shrubs, and smaller dicotyledonous herbs, as this environment provides optimum light requirements for *S. cinerea* to grow and eventually create dense canopies (Eser 1998; Partridge 1994). It grows at a wide soil fertility range from the more nutrient rich wetlands to peat bogs, with saline or high altitude sites beyond its limits (Partridge 1994).

Once established, *S. cinerea* are considered to have various impacts on indigenous wetland communities. Invasion can cause displacement of native plant communities as *S. cinerea* can outgrow these and ultimately establish dense canopies, modifying native understorey communities (Eser 1998; Partridge 1994; Thompson & Reeves 1994). Eser (1998) study in the South Taupo Wetland, found *S. cinerea* in areas of open water communities, and invaded vegetation among raupo reedlands, oioi (*Leptocarpus similis*) rushlands and *Baumea* sedgelands (Eser 1998).

A recent study by Coleman (2010) was undertaken to determine the coexistence of *S. cinerea* and *Dacrycarpus dacrydiodes* (kahikatea) in the Waikato Region. Kahikatea, an endemic podocarp, was once the major component of the Waikato swamp forests in the Waikato. The study demonstrated that *S. cinerea* inhibited the regeneration of kahikatea by surpassing growth to canopy, shading out further

recruitment, and maintaining dominance through proficient vegetative reproduction (Coleman 2010).

1.2.2 Willow Control

Historically, the importance of most willow management programs in New Zealand and Australia has been focused on chemical and manual control of infestations, often with minimal regard to the long term effects or ecological consequences of these practices (Williams & West 2000). In New Zealand, the Department of Conservation (DOC) administers the management of invasive weeds in natural areas throughout the country. DOC is responsible for managing environmental weeds in all protected natural areas it administers, which is approximately 30% of the total land area (Department of Conservation & New Zealand Conservation Authority 2006).

Various management techniques have been trialled to control willows in New Zealand wetlands (Husted-Andersen 2002). Intensive willow control programmes became the major option in an attempt to restore and maintain wetland vegetation types to their former condition. Current tools available for willow control in wetlands are restricted to mechanical control and chemical ground-based treatments, which often have limited success and various disadvantages. These methods include the cut and paint, drill and inject, aerial spot spray and aerial boom (Maguire 2010). The cut and paint method involves cutting the stems and painting the stumps with a herbicide such as glysophate or metsulfuron. The drill and inject method requires drilling holes into the sapwood and gel-formulated herbicide injected into the hole. Both of these methods are a combination of physical and chemical treatments that can be laborious, time consuming and expensive, as moving within wetland terrain can be sometimes difficult and contracting for large scale operations is costly (Husted-Andersen 2002). With the need for contractors in the wetland they can also cause unnecessary damage to native plant communities. Furthermore, trials have demonstrated that the stumps and roots may re-sprout and require further treatment (Husted-Andersen 2002). Aerial spot spray means spraying the individual plant with glyphosate solution from a helicopter using a hand-held spraying wand. This method requires high

costs due to contracting the light aircraft and trials showed that this technique did not kill large individuals as aerial spot spraying has difficulties reaching the meristem of the plant, in which glyphosate needs to target (Husted-Andersen 2002).

Most recently DOC engaged the National Institute of Water and Atmospheric Research (NIWA) to conduct trials of the herbicide Garlon®, to investigate successful control treatment of *S. cinerea* within New Zealand wetlands (Champion et al. 2008). Field trials of aerial application within the South Taupo Wetland were applied to an area of *S. cinerea* dominated vegetation. The outcome of the study demonstrated 95% control of *S. cinerea* with limited off-target damage to indigenous species.

1.3 Microfauna in freshwater wetlands

Emphasis on controlling *Salix* species throughout New Zealand wetlands has been based on willow kill rates, and restoring and maintaining native wetland vegetation types. Given the widespread distribution of willows within New Zealand waterways, it is surprising that very few studies have specifically examined their impacts on other aquatic life (Collier 1994).

Aquatic invertebrates are found in all freshwater systems, including rivers, lakes and wetlands. They inhabit the bottom substrate, swim in the water column, or live on the surface of the water (Suren & Sorrell 2010). They play a crucial role in transferring plant-based organic carbon into animal-based organic carbon, which is then available to fish and birds. They also have fundamental biodiversity and ecological values, as the majority are native to New Zealand, and many are endemic (Suren & Sorrell 2010).

The scope of research performed in New Zealand limits any conclusions regarding the impacts of willows on aquatic ecosystems, and has been limited to the effects on benthic macroinvertebrates (Collier 1994). Studies performed on streams and rivers in New Zealand have demonstrated that the density of willows can determine their ecological impact on aquatic ecology. Densely willow lined

sections can be detrimental to aquatic invertebrates whereas moderate plantings of riparian willow can improve aquatic invertebrate habitat conditions (Collier 1994; Lester et al. 1994; Glover & Sagar 1994). Collier (1994) found that potential effects of willows on aquatic invertebrate communities can include changes in physical habitat, water chemistry and changes in food supply (Collier 1994). Lester et al. (1994) observed lower aquatic invertebrate densities and biomass in densely willow-lined sections of the streams, which may have been a result of a decrease in average substrate size by reducing access to interstitial spaces between stones and/or inadequate food production through shade effects. Glover and Saga (1994) also found lower aquatic invertebrate densities and biomass in densely willow lined sections of streams compared to moderately willow-lined sections of the river (Glover & Sagar 1994). These findings were further demonstrated recently in 2011 on the Waikato River (Johnston 2011), indicating that densely lined willow sections consisted of lower aquatic invertebrate densities and biomass; however, a the combination of willow and other riparian plants supported a high diversity of aquatic invertebrates, suggesting that aquatic invertebrates preferred habitat heterogeneity. These studies focus on the relationship between benthic macroinvertebrates and willow densities in stream and river systems.

Another form of aquatic response can be seen from changes in microfaunal assemblages, as they are known to react to environmental modifications (Schindler 1987). These assemblages are sensitive to environmental conditions and respond promptly to any changes (Attayde & Bozelli 1998). Microfauna communities have been studied in deep and more stable environments such as lakes (Duggan et al. 2002), and very rarely in wetlands (Schoenberg 1988). Structure and dynamics of aquatic communities in wetlands are regulated by diverse and complex biotic and abiotic factors typical to the nature of wetland systems such as hydrologic fluctuations (Mitsch & Gosselink 2007), depth of water column, local climate and food web traits (Ortega-Mayagoitia et al. 2000). Microfauna are an important part of wetland foodwebs as they provide a vital link connecting primary producers of plants and algae to secondary consumers within the web such as fish and birds (Lougheed & Chow-Fraser 1998). Their major role within the food web is to consume detritus and convert into a food source for

bacteria and other microorganisms (Williams & Altmann 1980). Despite their key role, very little is known regarding microfaunal communities in wetland ecosystems. Lake studies suggest that microfaunal communities are structured by a variety of biotic components, such as food availability (Lougheed & Chow-Fraser 1998) and dispersal ability (Duggan et al. 2002), as well as biotic factors such as nutrient levels (Duggan et al. 2002), turbidity and temperature (Kirk & Gilbert 1990).

In aquatic ecosystems, changes in species composition have been considered as the earliest detectors of environmental stress (Schindler 1987). Despite their potential as effective indicators of environmental change, microfaunal assemblages have not been commonly used to measure the condition of an ecosystem (Attayde & Bozelli 1998). Microfauna assemblages of inland waters are a vital component of aquatic ecosystems. Any activity which impacts directly or indirectly in reducing composition or diversity may possibly remove them from the aquatic food web thus disturbing the survival of higher organisms reliant on this food source (Davis et al. 1997).

A study of microfaunal assemblages within a New Zealand wetland was performed in 2001, in the Whangamarino Wetland, south of Auckland (Ryan 2001). This study found diverse assemblages of microfauna including copepods and cladocerans. Ryan (2001) found that microfauna may be valuable as biological indicators to disturbance within wetlands as they respond quickly to environmental stress due to their short lifecycles and sensitivity to environmental change. The study also demonstrated that specific microfaunal assemblages correlated strongly with diverse vegetation classes symptomatic of disturbance gradients, such as areas of *Salix* species invading the wetland, associated with high temperature, pH and conductivity (Ryan 2001).

1.4 Research Aims & Objectives

The aim of this research is to quantitatively examine microfaunal assemblage abundance, richness and community composition among *Salix cinerea* stands within freshwater wetlands and determine whether these microfaunal assemblages are affected by willow growth and willow control treatment.

I hypothesise that microfaunal assemblage diversity and composition will differ between native wetland vegetation and *S. cinerea* stands due to differences in growth form and the deciduous nature of *S. cinerea*. My second hypothesis is that treatment of willow will lead to microfaunal diversity and composition becoming similar to that among native vegetation. Alternatively, willow treatment may have a negative effect on microfaunal assemblages.

Research questions to be addressed:

- What microfaunal species inhabit wetlands dominated by *S. cinerea*?
- What is the degree of variation in microfaunal community composition between native vegetation versus *S. cinerea* stands?
- To what degree does willow treatment affect the abundance and diversity of microfaunal assemblages

The objective of this research is to provide necessary quantitative data regarding microfaunal assemblage abundance, richness and community composition among *S. cinerea* stands within freshwater wetlands. A study of microfaunal assemblages living among *S. cinerea* stands in wetlands is timely and will be of immense value ecologically and economically to wetland managers in order to make more informed decisions regarding willow control treatment for the purposes of restoration initiatives of New Zealand's freshwater wetlands.

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CHAPTER TWO: METHODS

2.1 Study Area

The South Taupo Wetland, one of the largest wetlands in the North Island, New Zealand, is situated on the southern shores of Lake Taupo. Its area of 1500 ha extends from Motuoapa in the east to Waihi in the west and incorporates the Tongariro River Delta (Figure 2.1). Its ecological importance is highly regarded and has been acknowledged by the Department of Conservation (DOC) Tongariro-Taupo Conservancy since the late 1980s, due to supporting an exceptionally high diversity of flora and fauna (Cromarty & Scott 1995).

The wetland was formed following the last Taupo Eruption, around 1800 years ago, by the deposition of tephra onto the surrounding landscape, which was eroded and transported by the Tongariro, Waiotaka and Waimarino Rivers. This process formed the Tongariro Delta, many oxbows, and a series of beach ridges and hollows that run parallel to the lake edge, resulting in low lying waterlogged areas. Over the last 1800 years wetland vegetation has expanded and various permanently waterlogged areas have formed peat layers (Singers & Keys 2009).

The hydrology of the wetland is influenced by various factors, including regular flooding of the three main rivers, annual rainfall of approximately 1.2 m to 3.0 m (Cromarty & Scott 1995), groundwater from surrounding areas discharging into the wetland (Eser 1998), along with water level fluctuations from Lake Taupo. Lake Taupo water levels are artificially controlled for hydropower generation, which has resulted in a higher water table during the summer season (Singers & Keys 2009). In 1941 control gates were installed at the head of the Waikato River that resulted in lake levels exceeding 1 m above natural water levels in the first 10 years of operation (Eser 1998).

European colonisation of the Taupo Basin in the 1850s lead to the establishment of farming, and by 1941 large areas of manuka (*Leptospermum scoparium*) and kanuka (*Kunzea ericoides*) scrub/forest bordering the wetland had been cleared. The raising of the lake consequently inundated the operational farmland at the periphery of the wetland, becoming unsuitable for pasture, and allowed the

regeneration of a mosaic of native wetland along with the establishment of non-native plant species (Cromarty & Scott 1995).

The South Taupo Wetland is comprised of a complex mosaic of vegetation types including open, low stature plant communities, shrublands, low stature vegetation among scattered shrubs, and open water sections with submerged plants and emergent plant species lining the borders (Eser 1998). Some of these wetland habitats include raupo (*Typha orientalis*) reedland and giant spiked sedge (*Eleocharis sphacelata*) in permanently wet ponds, large areas of sedge (*Baumea rubiginosa*) rushland, oioi (*Leptocarpus similis*) rushland, flax (*Phormium tenax*) land and wetland scrub of mingimingi (*Coprosma propinqua*), twiggy tree daisy (*Oleria virgata*), ti kouka (*Cordyline australis*) and manuka (Eser 1998; Singers 2009).

The major current threat to the South Taupo Wetland is the rapid invasion of *Salix cinerea* and its displacement of indigenous wetland vegetation (Cromarty and Scott 1995; Eser 1998, DOC 2002; Singers 2009). *Salix cinerea* was first observed in the late 1970s, and by 1984 dense *S. cinerea* forest had covered 67.2 ha and a further 220.2 ha was colonised by young, scattered *S. cinerea* shrubs. By 1996, various densities of *S. cinerea* had covered a total area of 432 ha throughout the wetland (Eser 1998). This rapid spread of *S. cinerea* throughout the wetland is contributed by three main factors; 1) the location of established communities of *S. cinerea*, 2) the density of *S. cinerea* communities inhabiting the area, and 3) historical disturbances such as flood, fire or clearance of wetland vegetation (Eser 1998).

The Department of Conservation has recently written an Operational Plan (Department of Conservation 2010) in support of future willow control programmes throughout the South Taupo Wetland (Maguire 2010). There are a total of 10 wetland sites administered by DOC within the wetland. These consist of one Conservation Area, five Recreation Reserves and four Scenic Reserves, with a total area of approximately 485 ha (Jenkins 2007).

The Waimarino Recreation Reserve (83 ha) has received the most intensive willow control regimes; trials started in the summer season of 2005 and have continued every year with various control techniques and success rates, and will continue throughout the duration of the Operational Plan 2015 (Maguire 2010). The objectives of the initial trials were to evaluate treatment methods that best suited the level of willow infestation, cost effectiveness, efficacy, and impact and recovery on native vegetation (Singers 2007). These methods included ground control treatment such as drill and inject, and aerial control using aerial boom spray. The trials demonstrated that ground control treatment was effective when sites were easily accessible, and kill rates were high in areas of low willow infestation (<10-20% cover). However, as willow density became more substantial, ground control became less practical due to the effort required to treat an area. Nevertheless, the trial found that ground control is the most effective method for areas of rare and threatened dicot species. Various concentrations of metsulfuron herbicide mix were used for aerial boom spraying treatment.

Observations of aerial boom spraying showed that low concentrations of selected herbicide did not kill all willows, especially the tall and large trees. In the following summer season these areas were repeated with a higher concentration, and the kill rate was significantly greater (Singers 2007).

2.1.1 Waiotaka Scenic Reserve

The Waiotaka Scenic Reserve, also part of the South Taupo Wetland, is 29.18 ha in size, a low gradient wetland bordering the shore of Lake Taupo, created with beach ridges, pumice and greywacke alluvium. The hydrosystem is riverine influenced by the Waiotaka River floodplain. The reserve consists of ti kouka and kanuka forest on the dune ridges, sedge rushland (*Baumea rubiginosa*) peat bog, raupo reedland, manuka shubland, flaxland, toetoe (*Cortaderia toetoe*), tussockland and open water (Department of Conservation 2002). The reserve has been infested with a variety of exotic plants including *Salix cinerea*. DOC has included the Waiotaka Scenic Reserve into the Operational Plan and divided it into six operational blocks for weed management.

The site chosen for my study is the Waiotaka Scenic Reserve (Figure 2.4). The reserve consists of two blocks divided by a sandbar, which is known to DOC as Blocks 1 and 2. Block 1 is 8.4 ha and situated parallel to State Highway One. Ground control of *S. cinerea* took place in Block 1 (8.4 ha) in summer 2007/2008 using a variety of methods, including vehicle mounted spraying, cut and gel, and drill and inject. Block 2 is 6.3 ha and located closest to the lake shore and blocked by the boat access and car park administered by the Taupo District Council. Block 2 has received no willow control prior to this study. Access to Blocks 1 and 2 is from State Highway 1 and onto Frethey Drive.

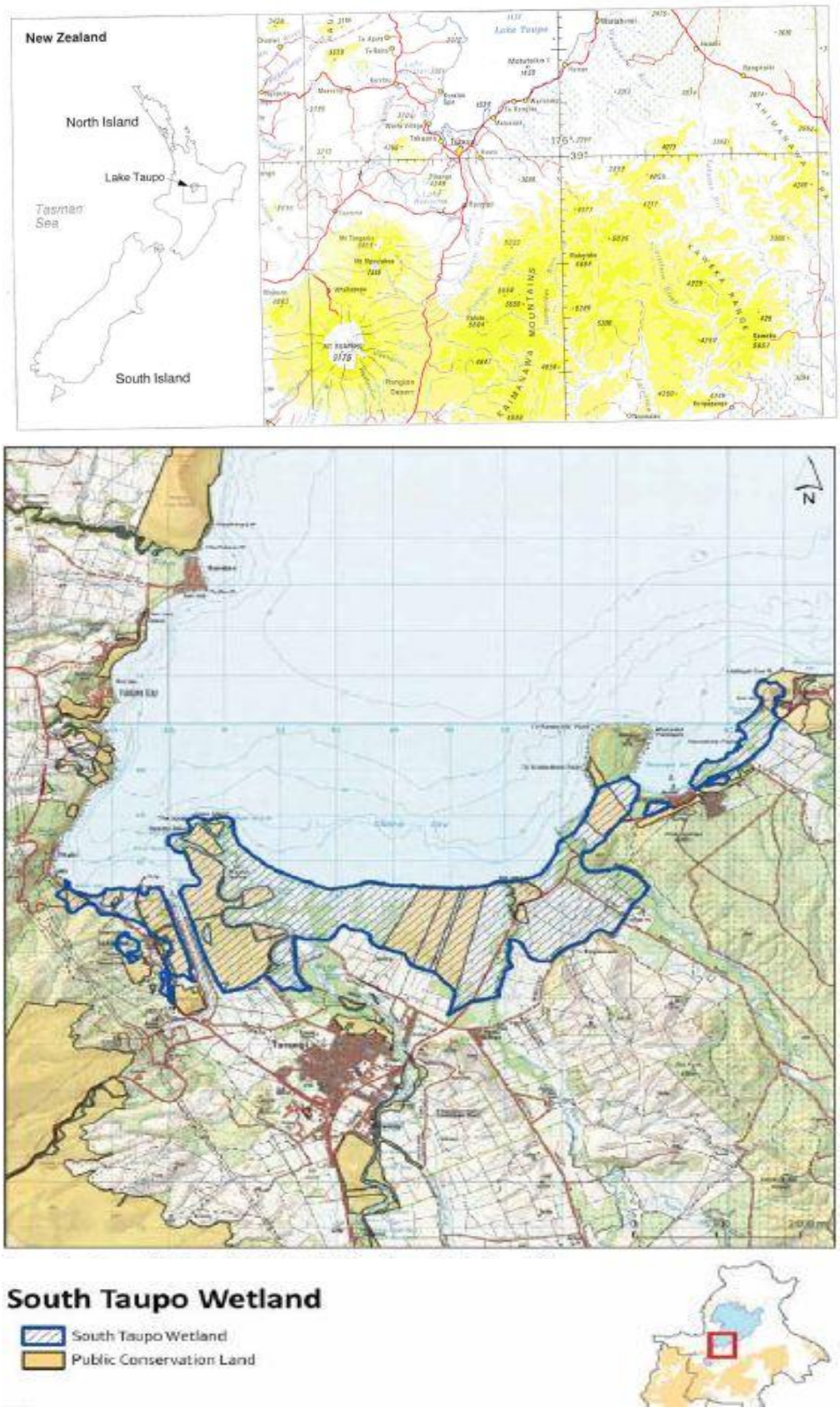

Figure 2.1 Map of South Taupo Wetland, Central North Island, New Zealand
Maps courtesy of Eser (1998) and Singers (2009)

2.2 Sampling Sites

2.2.1 Long-term effects of microfaunal community composition between native vegetation versus *S. cinerea*

Seven native sites (N1-N7) were chosen in both Block 1 (Figure 2.2) and 2 (Figure 2.3) that represented indigenous wetland plant species that is not encroached by willow. These sites consisted of raupo (*Typha orientalis*), *Cortaderia toetoe* and sedges including *Baumea rubiginosa* and *Carex secta*, and open water. This mix of native vegetation reflects the most favourable communities for *S. cinerea* to potentially invade. Seven native sites were selected based on permanently wet areas and located close to live and dead *S. cinerea* trees. Live *Salix cinerea* sites (L1-L7) were chosen in Block 2 invaded by *S. cinerea* that had never been treated. Seven living *S. cinerea* individual trees taller than 2 m, scattered throughout the block and located in permanently wet areas were selected for this experiment. Dead *Salix cinerea* sites (D1-D7) were chosen in Block 1, which contained *S. cinerea* that were treated in summer season 2007/2008. Seven dead *S. cinerea* individual trees taller than 2 m, scattered throughout the block and located in permanently wet areas, were selected for this experiment.

Sampling was undertaken in February (late summer), July (winter) and December (early summer) 2011, to encompass seasonal variation. During these times *Salix cinerea* was in late summer bloom, had lost their leaves (winter), or were in early summer bloom, respectively. Sampling was undertaken by wading, with the wetland accessible during high water depths using chest waders.

Figure 2.2 Aerial photograph of Block One (Dead *Salix cinerea*) Courtesy of DOC Turangi Office 25/11/2010

Figure 2.3 Aerial photograph of Block Two (Live *Salix cinerea*) Courtesy of DOC Turangi Office 25/11/2010

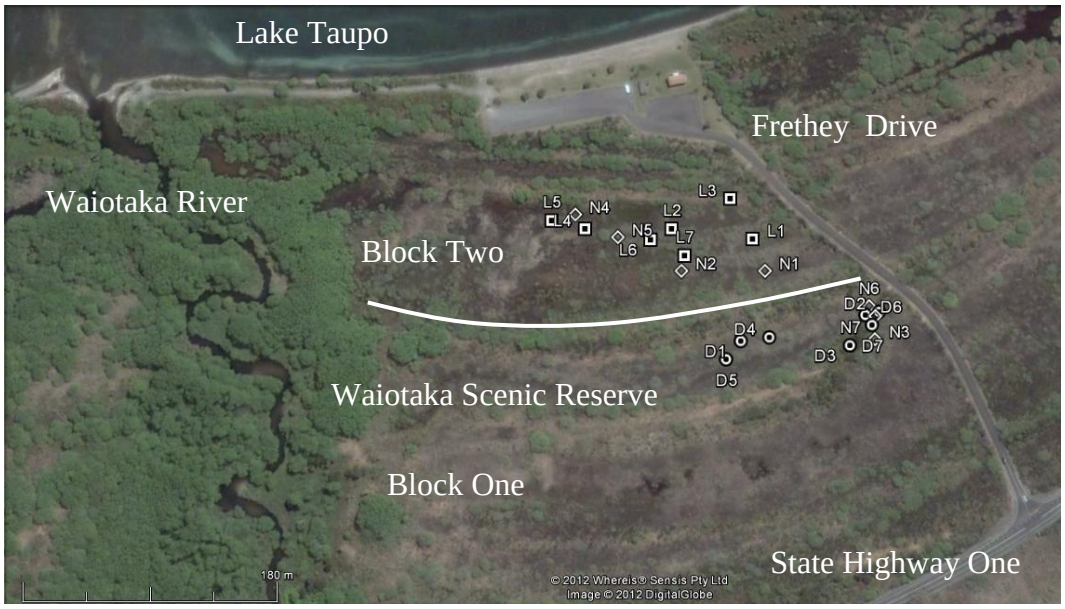

Figure 2.4 Waiotaka Scenic Reserve illustrating the native, live willow and dead willow in Blocks 1 & 2. Salix cinerea were poisoned in Block in the summer season of 2007/2008

2.2.2 Short-term effects of microfaunal community composition post willow control treatment

Eight live *Salix cinerea* trees (L5 and L6 from the previous experiment, plus L8-L13) were sampled on 1 February 2012(Figure 2.5). Ground control using the drill and inject method took place on the 16 February 2012. The Department of Conservation Turangi Conservancy used trained contractors to treat seven *S. cinerea* individuals. Each tree was treated by drilling approximately 100 mm into the stems with a wood auger bit (20 mm diameter) at approximately 100 mm spacing around the trunk. Each hole was filled with 0.2 g/mL of metsulfuron solution. Each tree was drilled according to the size of stem and height of tree. Post-treatment sampling took place 14 March 2012, with six live *S. cinerea* (L1, L3, L5, L6, L8, L13) and seven treated *S. cinerea* (P1-P7) sampled.

Figure 2.5 Waiotaka Scenic Reserve illustrating the *Salix cinerea* control and treatment before and after

2.3 Physiochemical variables

Temperature, dissolved oxygen and specific conductance and pH were measured at each sampling site, using YSI 85 and Oakton Waterproof pHTestr10 meters. Water depth was measured with a wooden ruler from the substrate to the surface water. Canopy cover of dead and living *Salix cinerea* was measured using a Spherical Densiometer Model A instrument by noting whether overhead shade occurred on each of the 25 squares as described in Harding et al. (Harding et al.

2009). Ground cover vegetation of native, live and dead *S. cinerea* was performed by estimating the abundant plant species of each site.

Chlorophyll *a* was collected by filtering 60 ml of undisturbed water from the water column at each site, through a 0.45 µm glass fiber microfilter. The filter was then folded in half, wrapped in aluminium foil and stored immediately on ice until returned the laboratory, where it was stored frozen in the dark until analysis. To extract and measure chlorophyll *a* the method of Arar and Collins (1997) was followed. Each filter paper was steeped in 90% acetone solution (buffered with magnesium carbonate) for 12 hours. Samples were then centrifuged and measured for fluorescence using a 10-AU fluorometer calibrated for chlorophyll *a* analysis.

2.4 Microfaunal sampling, enumeration and identification

10 L of undisturbed water was collected using a 2 L plastic jug from each site and filtered through a 40 µm mesh. Each sample was placed in a 250 ml container and immediately filled with 95% ethanol to attain a final concentration of at least 50% ethanol for preservation. Each pot was appropriately labelled with site number and date. In the laboratory, microfauna samples were diluted to a known volume dependant on the amount of sediment and detrital matter in the sample, to facilitate ease of counting. Subsamples of 5 ml were removed using an autopipette, placed in an open-topped Perspex counting tray (50 mm x 80 mm) on a moveable microscope stage and enumerated using a Nikon SM2800 stereo microscope. Successive subsamples were counted until 300 individuals were obtained, or until the entire sample was counted if less were encountered. Species were identified primarily using Chapman & Lewis (1976) and Shiel (1995) using an Olympus BX50 microscope.

2.5 Statistical Data Analysis

Multi-dimensional scaling (MDS) and analysis of similarities (ANOSIM) were used to identify patterns in community composition and to determine which environmental variables were associated with underlying trends in species distribution. MDS is a multivariate ordination technique that builds a 2D map of samples based on their similarity to each other as defined by a distance metric. A

stress value measures the quality of the map's fit, where zero indicates a perfect fit. The stress value indicates the degree of similarity and assesses the correlation between the distances of points on the MDS map and the distances in the original distance matrix. This technique is appropriate for data with non-normal distribution and presents results that are simple to interpret. MDS was achieved on the ranked Bray-Curtis similarity matrix (Clarke & Warwick 1994), and was calculated on log (x+1) transformed abundance data of common taxa using the PRIMER 6.0 statistical package (Plymouth Marine Laboratory). The log (x+1) transformation was chosen to reduce the influence of highly abundant taxa and increase the influence of important, but less abundant, community members. Rare taxa were removed from the analysis in order to remove the influence of species potentially sampled by chance. Common taxa were defined as those comprising >2 or more samples found in the sampling season. As sub-adult stages of cyclopoid copepods were difficult to assign to species, they were therefore treated as a single group in the analyses. Adult female cyclopoid copepods carrying egg sacs were identified from each sample.

To determine the effects of willow growth and willow control on microfaunal community composition, sample periods were divided a priori into three groups (February, July and December 2011). Associations between sample distribution and environmental data (habitat type) in each sample period were investigated firstly by superimposing environmental variables onto the ordinations. ANOSIM was then undertaken on the similarity matrix to test whether the community differences observed between the habitat types were statistically significant. ANOSIM is a non-parametric permutation test used with multivariate data to test a priori hypotheses (Clarke & Warwick 1994). The analysis provides a measure of the dissimilarity of groups of samples shown by an R-statistic that usually lies between 0 and 1. Values close to one indicates that the groups are dissimilar and those closest to zero demonstrate that groups are similar. ANOSIM was executed on the log (x+1) transformed Bray-Curtis dissimilarity matrices to test for significance of the effects of habitat and sample composition.

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CHAPTER THREE: LONG-TERM EFFECTS OF MICROFAUNAL COMPOSITION BETWEEN NATIVE VEGETATION *VERSUS* LIVE AND DEAD *SALIX CINEREA*

3.1 Results

3.1.1 Environmental Variables

In February, water depth (Figure 3.1) within the wetland averaged 19.2 cm among the native vegetation, 14.8 cm under dead willow, and 19.4 cm under live willow. However, in July water depth had risen in the native sites to 21.5 cm, live willow was lowest at 12.3 cm and dead willow had increased to 16.2 cm. Similarly, in December water depth in native vegetation averaged 18.6 cm, while live willow was lowest at 9.0 cm and dead willow averaged 15.4 cm. However, ANOVA indicated there was no significant difference in water depth between native, live and dead willow in any season ($P>0.05$; Table 3.1). Lake Taupo water levels (Figure 3.3) were at its highest in February, and gradually declined by July, with a slight increase by December. Rainfall (Figure 3.2) was lowest in February, increased by July and declined by December.

The average water temperature (Figure 3.4) was highest in February. Temperature among native vegetation and dead willow averaged 19.7˚C and live willow 19.2˚C. In July native and dead willow averaged 6.5˚C and live willow 7.3˚C, while in December the native vegetation averaged 18.1˚C, dead willow 17.3˚C and live willow 16.7˚C. ANOVA indicated there was no significant difference of water temperature between native, live and dead willows in any season ($P>0.05$; Table 3.2). The average pH (Figure 3.5) during February was 6.3 for native, 6.4 for live willow and 6.5 for dead willow. In July pH increased slightly to 6.7 and in December pH decreased to 6.5 for all vegetation types. ANOVA showed no significant difference in pH between native, live and dead willows in any season ($P>0.05$; Table 3.3). The average dissolved oxygen concentrations (Figure 3.6) were generally higher among dead willow, ranging from 0.6-1.8 mg/L throughout each season, while live willow was comparatively lower ranging from 0.2-0.5 mg/L and native ranging from 0.3-0.8 mg/L. ANOVA indicated there was a

significant difference in dissolved oxygen among sites in February (P value=0.030; Table 3.4); the Tukey post-hoc test indicated dissolved oxygen was significantly lower in the live willow sites than among dead willows (P value=0.030; Table 3.5). However, there were no significant differences between native, live and dead willows in July and December (P>0.05; Table 3.4). The average specific conductance (Figure 3.7) was highest in February, ranging between 184.2 and 230.1 µS/cm, decreasing in July to between 108.6 and 153.3 µS/cm, and again in December to 112.3 and 141.9 µS/cm. ANOVA indicated there was no significant difference in specific conductance between native, live and dead willows in any season (P>0.05; Table 3.6). In February, the average chlorophyll *a* concentration (Figure 3.8) was highest among live willow at 2.5 µg/L, and lower in native at 2.3 µg/L and dead willow at 1.3 µg/L. In July chlorophyll *a* concentration in dead willow was highest at 3.9 µg/L, live willow 1.6 µg/L and native 1.0 µg/L. In December average chlorophyll *a* was highest among live willow at 4.1 µg/L, native 3.4 µg/L and lowest among dead willow at 0.8 µg/L. ANOVA indicated there were no significant differences in chlorophyll *a* between sites for any season (P>0.05; Table 3.7). Average canopy density (Figure 3.9) was measured among live and dead willow only as native did not have canopy cover. Canopy density was highest among live willow during each season compared to dead willow. T-test results indicated there was a significant difference in canopy density between live and dead willows in each season (P≤0.05; Table 3.8). Average vegetation ground cover (Figure 3.10) for native sites consisted of 75% *Baumea rubiginosa* and 20% open water, with a mix of *Carex secta, Typha orientalis* and *Cortaderia toetoe*. Live willows were surrounded by 40% *B. rubiginosa* and 31% open water with a combination of *C. secta, C. toetoe, Leptocarpus similis, Coprosma robusta* and *T. orientalis*. Dead willows were surrounded by 44% open water and 27% *B. rubiginosa,* with *L. similis, C. robusta, Phormium tenax* and young shoots of *Salix cinerea.*

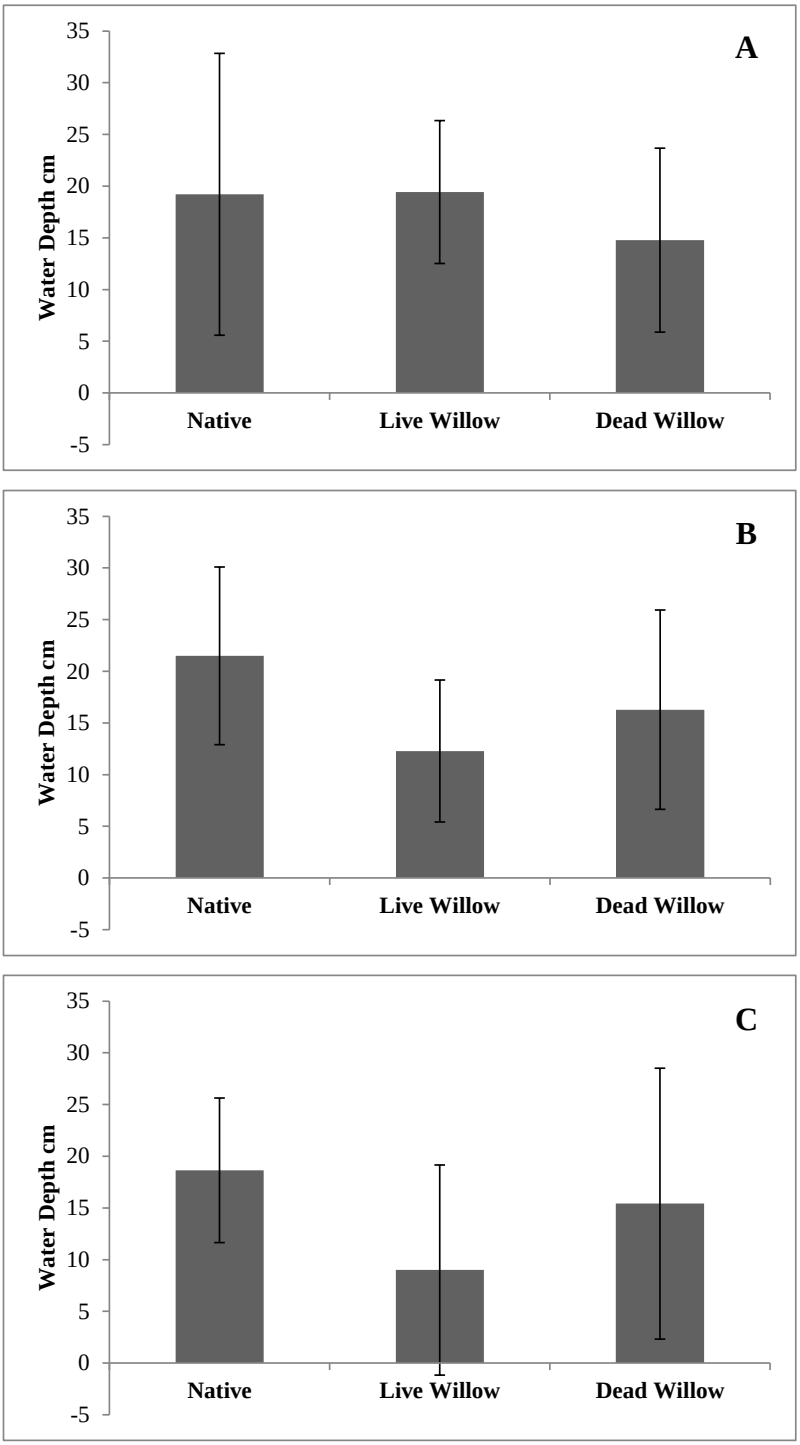


Figure 3.1 Mean±SD water depth of native, live willow and dead willow for A) February, B) July & C) December

Table 3.1 Summary of ANOVA results for mean water depth for February, July and December showing sum of squares (SS), degree of freedom (DF), mean squares (MS), F values (F) and probability values (P)

Water Depth cm	SS	DF	MS	F	P
February	0.099	2	0.049	0.841	0.447
July	0.399	2	0.199	1.962	0.169
December	0.851	2	0.425	2.008	0.171

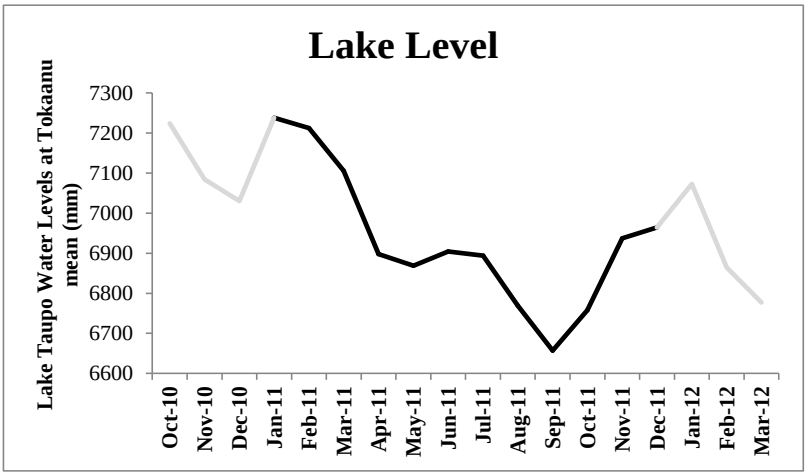


Figure 3.3 Lake Taupo at Tokaanu mean monthly lake levels from October 2010 – March 2012. Sample period between February 2011 – December 2011.
Data from NIWA, Tokaanu

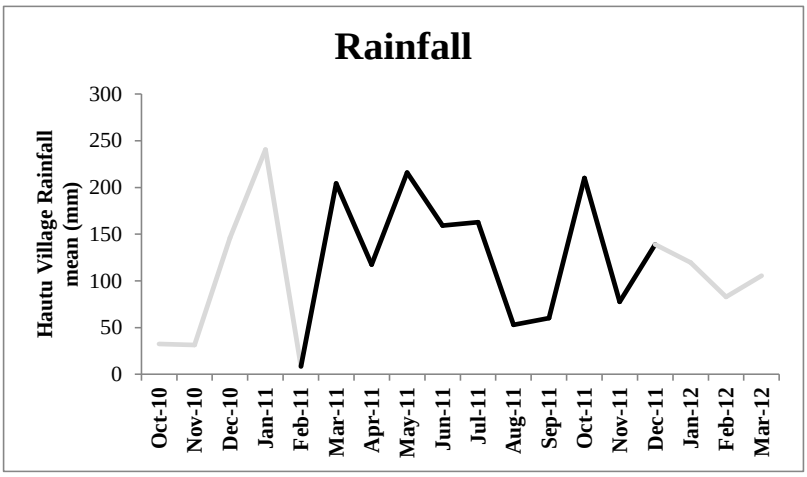


Figure 3.2 Hautu Village Station mean monthly rainfall from October 2010 – March 2012. Sample period between February 2011 – December 2011.
Data from http://cliflo.niwa.co.nz/pls/niwp/doc/terms.html

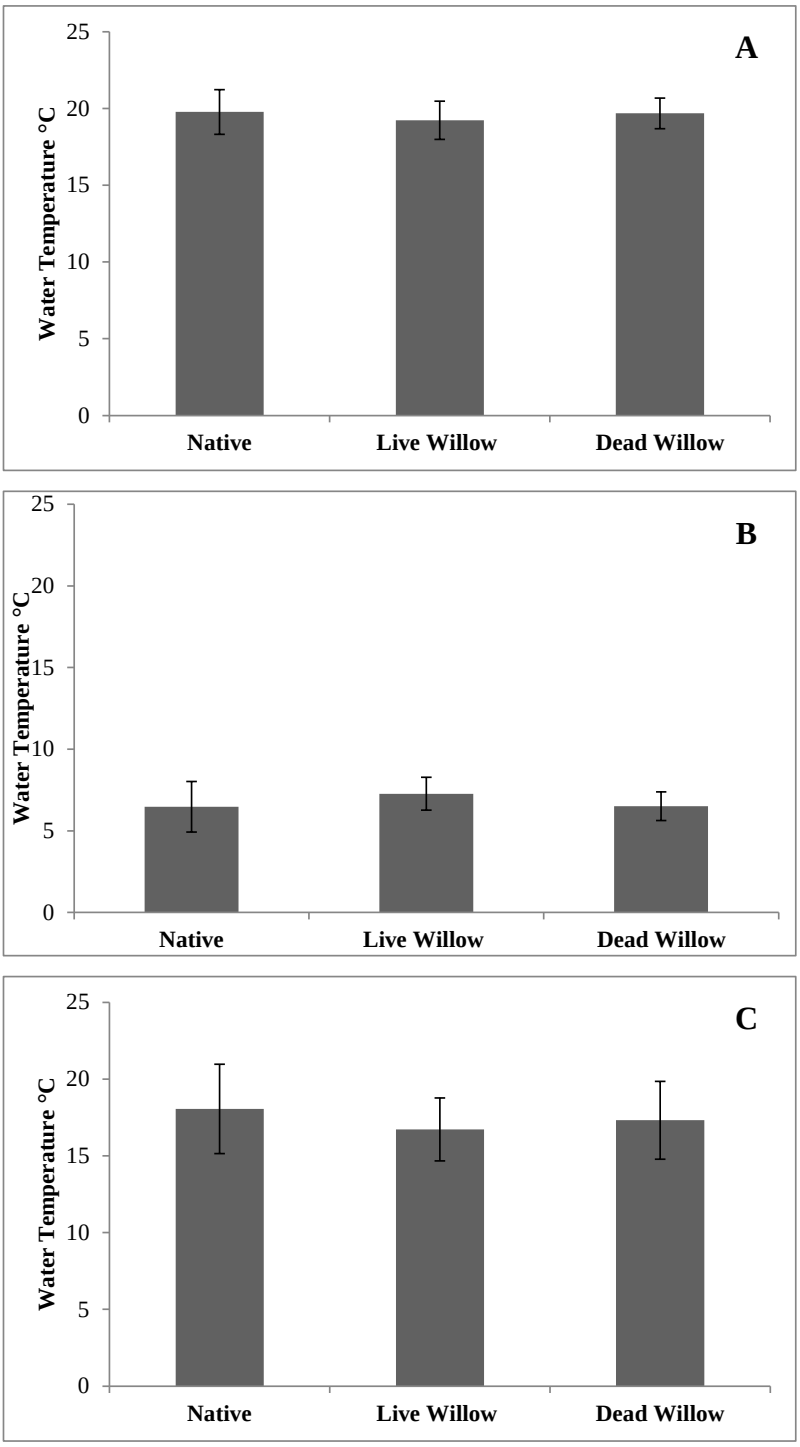


Figure 3.4 Mean±SD water temperature of native, live willow and dead willow for A) February, B) July & C) December

Table 3.2 Summary of ANOVA results for mean water temperature for February, July and December showing sum of squares (SS), degree of freedom (DF), mean squares (MS), F values (F) and probability values (P)

Water Temperature °C	SS	DF	MS	F	P
February	0.001	2	0.000	0.384	0.687
July	0.013	2	0.007	0.991	0.391
December	0.002	2	0.001	0.293	0.750

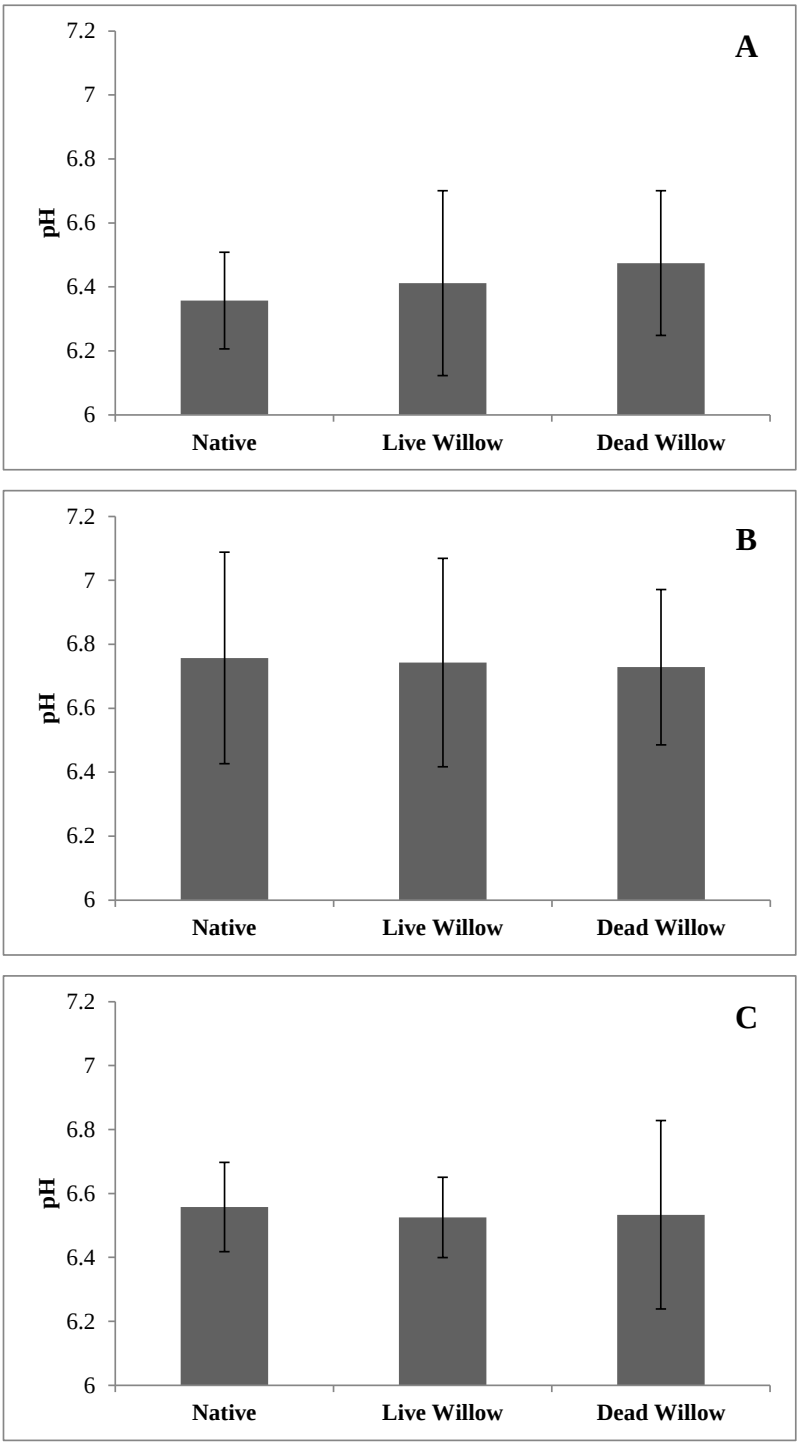


Figure 3.5 Mean±SD pH of native, live willow and dead willow for A) February, B) July & C) December

Table 3.3 Summary of ANOVA results for mean pH for February, July and December showing sum of squares (SS), degree of freedom (DF), mean squares (MS), F values (F) and probability values (P)

pH	SS	DF	MS	F	P
February	0.000	2	0.000	0.455	0.642
July	0.000	2	0.000	0.012	0.988
December	0.000	2	0.000	0.042	0.959

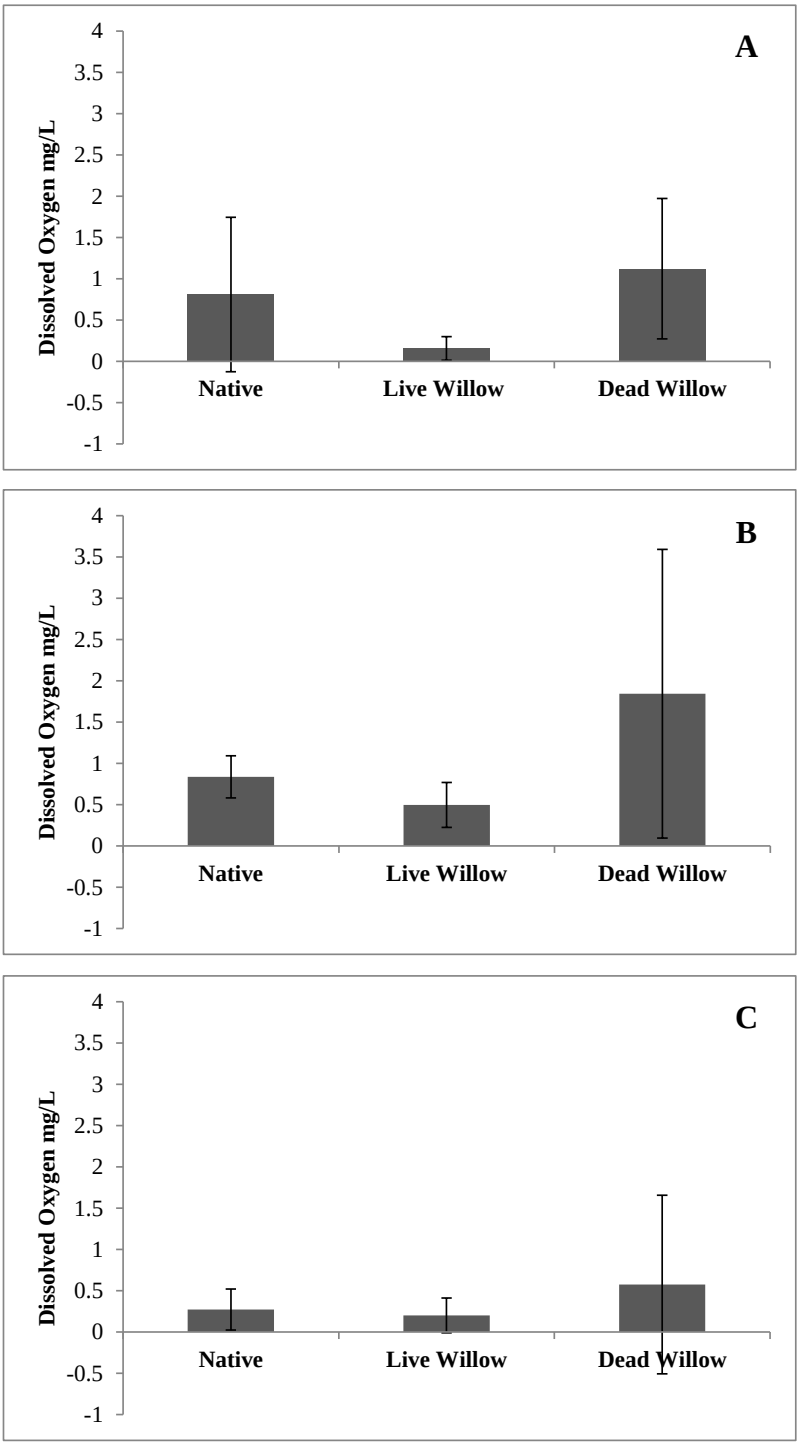


Figure 3.6 Mean±SD dissolved oxygen of native, live willow and dead willow for A) February, B) July & C) December

Table 3.4 Summary of ANOVA results for mean dissolved oxygen for February, July and December showing sum of squares (SS), degree of freedom (DF), mean squares (MS), F values (F) and probability values (P). * indicates significance

Dissolved Oxygen mg/L	SS	DF	MS	F	P
February	0.202	2	0.101	4.308	0.030*
July	0.958	2	0.479	2.967	0.077
December	0.222	2	0.111	0.296	0.749

Table 3.5 Post-Hoc Tukey Test results for mean dissolved oxygen for February.
*** indicates significance**

Dissolved Oxygen	P Value
Native/Live Willow	0.160
Native/Dead Willow	0.600
Live Willow/Dead Willow	0.030*

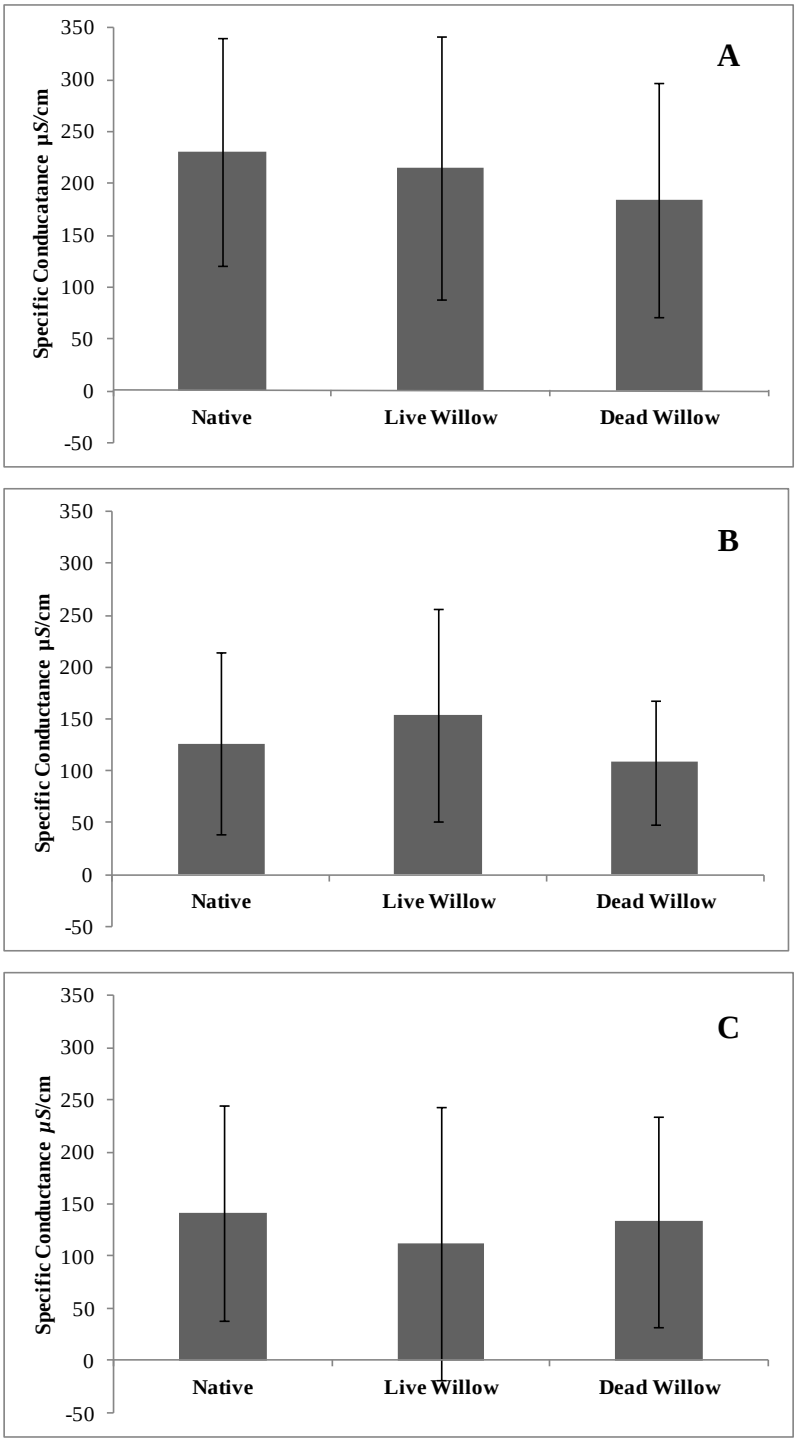


Figure 3.7 Mean±SD specific conductance of native, live willow and dead willow for A) February, B) July & C) December

Table 3.6 Summary of ANOVA results for mean specific conductance for February, July and December showing sum of squares (SS), degree of freedom (DF), mean squares (MS), F values (F) and probability values (P)

Specific Conducttance μS/cm	SS	DF	MS	F	P
February	0.045	2	0.022	0.462	0.637
July	0.076	2	0.038	0.085	0.918
December	3.238	2	1.619	0.674	0.525

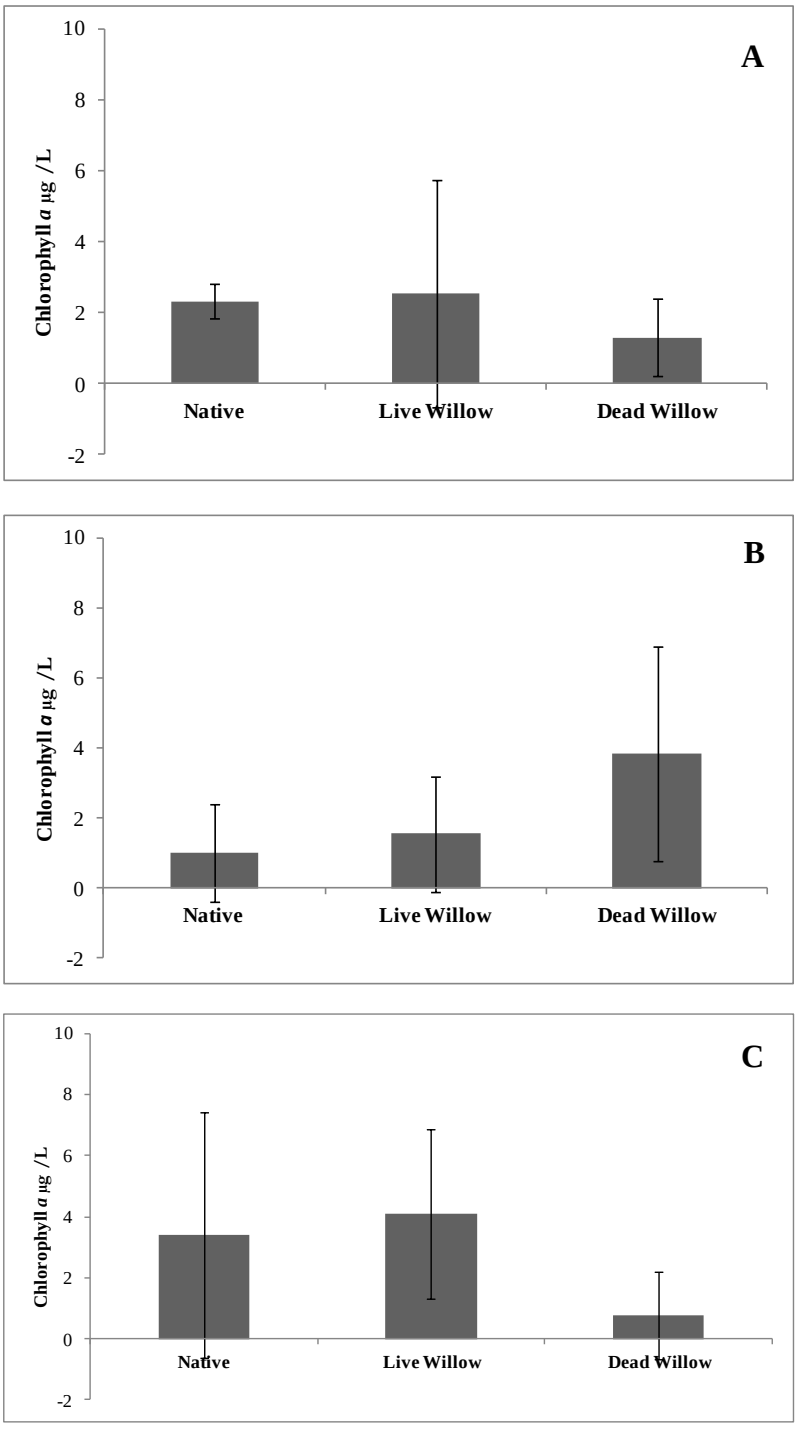


Figure 3.8 Mean±SD chlorophyll *a* of native, live willow and dead willow for A) February, B) July & C) December

Table 3.7 Summary of ANOVA results for mean chlorophyll *a* for February, July and December showing sum of squares (SS), degree of freedom (DF), mean squares (MS), F values (F) and probability values (P)

Chlorophyll *a* µg/L	SS	DF	MS	F	P
February	0.560	2	0.280	1.088	0.358
July	0.105	2	0.053	0.138	0.872
December	0.473	2	0.237	0.634	0.545

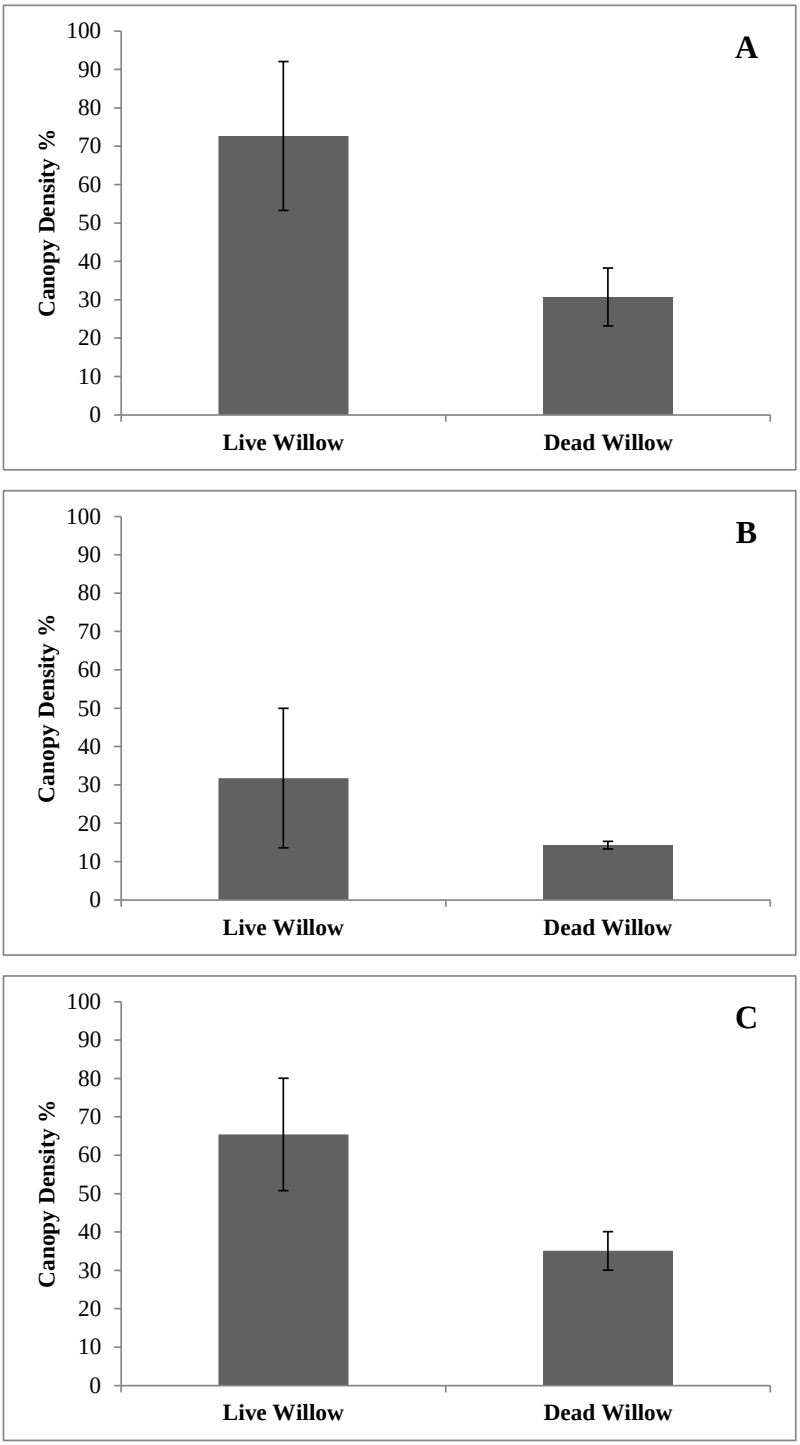


Figure 3.9 Mean±SD canopy density of native, live willow and dead willow for A) February, B) July & C) December

Table 3.8 Summary of T-test results for mean canopy density for before and after treatment showing t-values and probability values (P)

Canopy Density	t-value	P
February	6.014	0.000
July	3.416	0.005
December	4.971	0.002

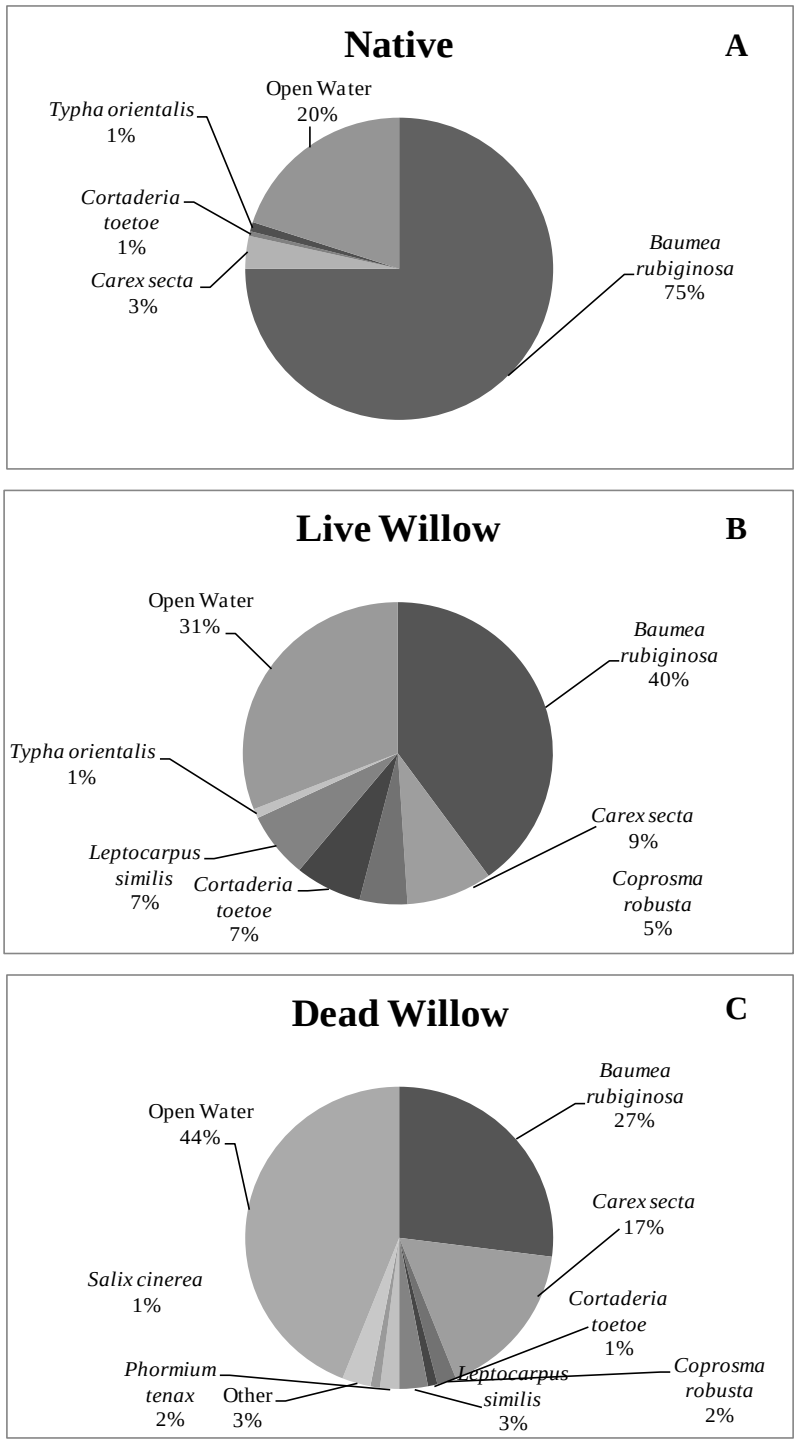


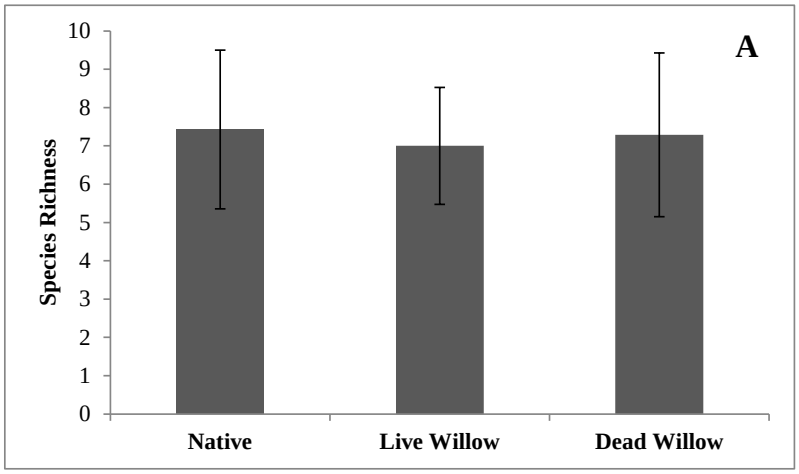
Figure 3.10 Average percentage of vegetation ground cover for A) native, B) live willow and C) dead willow

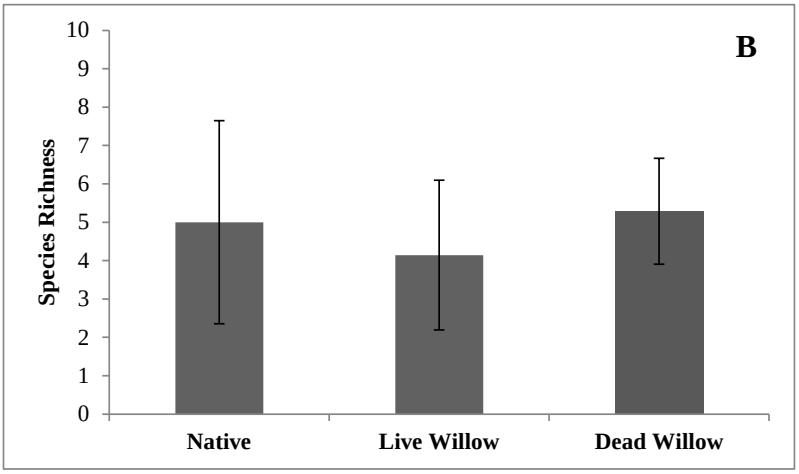
3.1.2 Microfaunal Composition and Dynamics

Species richness was highest in February (Figure 3.11A) with an average of seven species found among native, live and dead willows. The major taxa recorded for February consisted of copepods, including cyclopoid copepods such as *Acanthocyclops robustus*, *Tropocyclops prisinus*, cyclopoid nauplii and the harpacticoid copepod *Attheyella lewisae*, and cladocerans such as *Chydorus* sp., *Ceriodaphnia dubia* and *Simocephalus vetulus*, and ostracods. Cyclopoid copepods were the dominant species in native, live willow and dead willow sites (Figure 3.12A).The most diverse taxa (Table 3.11) were rotifers. These consisted of bdelloids, *Lecane bulla*, *L. closterocerca*, *L. hamata*, *L. lunaris*, *Notomatta allontois*, *Polyarthra dolichoptera*, *Proales decipiens*, *Scaridium longicaudum*, *Trichocerca similis*, *Trichocerca* sp. and *Trichotria tetractis*. The major taxa recorded for July consisted of copepods, including *A. robustus*, *T. prisinus*, *Diacyclops bicuspidatus*, *Mesocyclops* sp., cyclopoid nauplii and the harpacticoid cyclopod *A. lewisae*, and cladocerans including *Chydorus* sp., *C. dubia* and *S. vetulus*, and ostracods. Ostracods were the most abundant taxa in July found among live willow (Figure 3.12B). In July species richness was lower (Figure 3.11B), averaging four species in live willow, and five in native and dead willow. Rotifers were the diverse taxa for July (Table 3.11), these consisted of bdelloids, *Cupelopagis vorax*, *Lecane pusilla*, *N. allontois*, *P. decipiens*, *S. longicaudum*, *Squatinella mutica*, *T. similis*, *Trichocerca tigris* and *T. tetractis*. Fewer species were found in December, ranging from three in live willow, four in native and five in dead willow sites (Figure 3.11C). The major taxa recorded for December were copepods, including cyclopoid copepods such as *A. robustus*, *T. prisinus*, *Eucyclops serralatus* and cyclopoid nauplii, cladocerans such as *Chydorus* sp. and *S. vetulus*, and ostracods. Ostracods were the dominant taxa in December found among dead willow (Figure 3.12C). Rotifer diversity was lower in December compared to the previous months (Table 3.12). These included *Aspelta angusta*, *P. decipiens*, *S. longicaudum*, *Tetrasiphon hydrocora* and *Trichocerca* sp.

ANOVA indicated there were no significant differences in species richness between native, live and dead willow sites in any season (Table 3.9) MDS ordinations may be used to examine how similar the species composition of samples is to other samples. Samples that have similar species composition are

placed closer together. The samples formed two main clusters during each season. The MDS ordination for February represents two main clusters, live willow and natives sampled in Block 2 and dead willow and natives sampled in Block 1 (Figure 3.13A & 3.14A). ANOSIM results for native, live and dead willows shows no significant difference among the vegetation types (P>0.05; Table 3.14). However, ANOSIM results for the vegetation in Blocks 1 and 2 illustrates a significant difference (P value=0.024; Table 3.13). Similarly, the MDS ordination for July follows the same trend, with two main clustered groups (Figure 3.13B & Figure 3.14B). ANOSIM results for native, live and dead willows show no significant difference among the vegetation types (P>0.05; Table 3.13). However, ANOSIM results for Blocks 1 and 2 reveal a significant difference (P value=0.012; Table 3.14). MDS ordination for December showed two main clusters (Figure 3.13C & Figure 3.14C). ANOSIM results for native, live and dead willows revealed no significant difference among the vegetation types (P>0.05; Table 3.13), and no clear significant difference between Blocks 1 and 2 (P>0.05; Table 3.14).





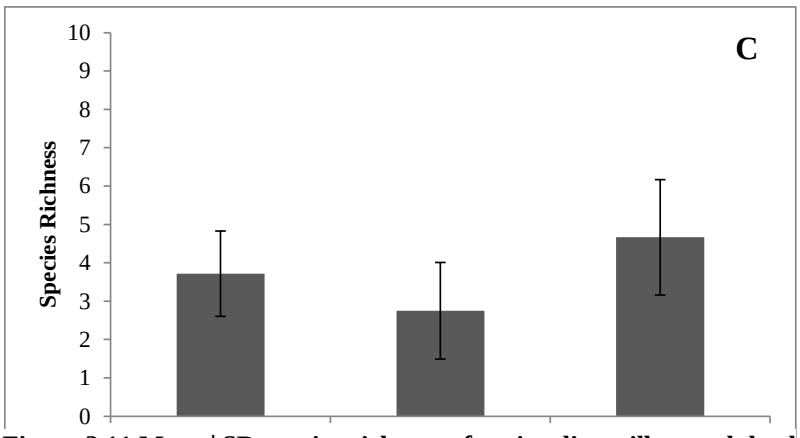


Figure 3.11 Mean±SD species richness of native, live willow and dead willow for A) February, B) July & C) December

Table 3.9 Summary of ANOVA results for mean species richness for February, July and December showing sum of squares (SS), degree of freedom (D.F), mean squares (MS), F values (F) and probability values (P)

Species Richness	SS	D. F	MS	F	P
February	0.667	2	0.333	0.089	0.915
July	4.952	2	2.476	0.584	0.568
December	4.952	2	2.476	0.584	0.568

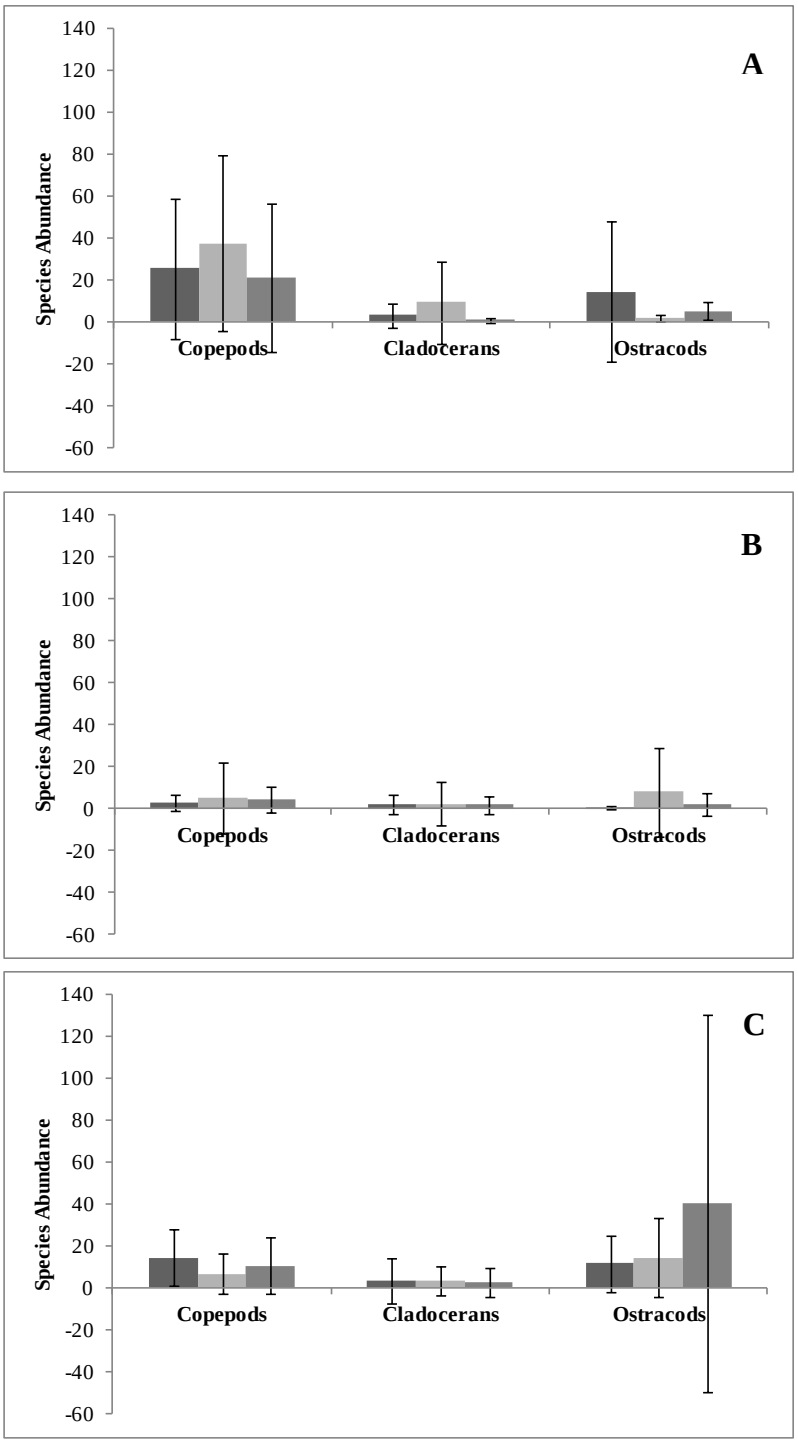


Figure 3.12 Mean±SD species abundance for native, live willow and dead willow for A) February, B) July & C) December

Table 3.10 Microfauna recorded in all sites during February 2011

Species	N1	N2	N3	N4	N5	N6	N7	L1	L2	L3	L4	L5	L6	L7	D1	D2	D3	D4	D5	D6	D7
Rotifera																					
Bdelloids	*			*						*	*						*				
Aspelta angusta Harring & Myers, 1928																					
Cupelopagis vorax Leidy, 1857																					
Lecane bulla Gosse, 1851					*						*	*		*							
L. closterocerca Schmarda, 1859				*													*				
L. hamata Stokes, 1896												*									
L. lunaris Ehrenbrg, 1832					*																
L. pusilla Harring, 1914																					
Notommata allantois Wulfert, 1935			*	*							*	*				*					
Polyarthra dolichoptera Idelson 1925												*									
Proales decipiens Ehrenberg, 1832																					*
Scaridium longicaudum Müller, 1786								*				*									
Squatinella mutica Ehrenberg, 1832																					
Tetrasiphon hydrocora Ehrenberg, 1840				*														*			
Trichocerca similis Wierzejski, 1893																					
T. tigris Müller, 1786																					
Trichocera sp.									*												
Trichotria tetractis Ehrenberg, 1830			*														*				
Cladocera																					
Alona guttata Sars, 1862							*							*							
A. quadrangularis Müller, 1776								*													
Camptocercus australis Sars, 1896																					
Ceriodaphnia dubia Richard, 1894		*		*		*		*	*									*			
Chydorus sp. Müller, 1785	*	*	*	*	*	*	*	*	*	*	*	*	*		*	*	*	*	*		*
Ilyocryptus sordidus Lièvin, 1848																					
Oxyurella tenuicaudis Sars, 1862														*							
Simocephalus vetulus Müller, 1776	*	*	*	*	*		*	*		*	*	*	*	*	*	*	*	*	*	*	
Copepoda																					
Cyclopoid copepod	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Acanthocyclops robustus Sars, 1863	*	*	*	*	*	*	*	*	*	*		*		*			*				
Diacyclops bicuspidatus Claus, 1857																					
Eucyclops serrulatus Fischer, 1851																					
Mesocyclops sp.																					
Paracyclops fimbriatus Fischer, 1853																					
Tropocyclops prasinus Fischer, 1860											*	*	*		*	*	*	*	*	*	*
Cyclopoid nauplii	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Harpacticoid Copepod																					
Attheyella lewisae Wells, 2007	*							*		*											*
Harpacticoid nauplii																					
Tardigrades				*												*	*				
Oligochaetes	*						*		*								*				
Ostracods	*	*	*		*	*	*	*		*	*	*	*	*	*	*	*	*	*	*	*
Chronomids			*		*			*				*			*	*	*			*	
Springtails	*		*	*			*								*		*	*		*	
Mites	*		*	*			*			*	*	*			*	*		*		*	
Gastrotrichs				*																	

Table 3.11 Microfauna recorded in all sites during July 2011

Species	N1	N2	N3	N4	N5	N6	N7	L1	L2	L3	L4	L5	L6	L7	D1	D2	D3	D4	D5	D6	D7
Rotifera																					
Bdelloids													*								
Aspelta angusta Harring & Myers, 1928																					
Cupelopagis vorax Leidy, 1857						*															
Lecane bulla Gosse, 1851																					
L. closterocerca Schmarda, 1859																					
L. hamata Stokes, 1896																					
L. lunaris Ehrenbrg, 1832																					
L. pusilla Harring, 1914						*															
Notommata allantois Wulfert, 1935																				*	
Polyarthra dolichoptera Idelson 1925																					
Proales decipiens Ehrenberg, 1832							*					*									
Scaridium longicaudum Müller, 1786				*							*	*									
Squatinella mutica Ehrenberg, 1832						*															
Tetrasiphon hydrocora Ehrenberg, 1840																	*				
Trichocerca similis Wierzejski, 1893																				*	
T. tigris Müller, 1786							*														
Trichocera sp.																					
Trichotria tetractis Ehrenberg, 1830			*													*					
Cladocera																					
Alona guttata Sars, 1862																					
A. quadrangularis Müller, 1776																					
Camptocercus australis Sars, 1896							*											*			
Ceriodaphnia dubia Richard, 1894						*	*									*	*			*	
Chydorus sp. Müller, 1785			*		*	*	*	*		*		*	*		*	*	*	*		*	*
Ilyocryptus sordidus Lièvin, 1848				*																	
Oxyurella tenuicaudis Sars, 1862																					
Simocephalus vetulus Müller, 1776			*			*	*										*		*	*	*
Copepoda																					
Cyclopoid copepod	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Acanthocyclops robustus Sars, 1863	*	*	*		*										*	*	*	*	*	*	*
Diacyclops bicuspidatus Claus, 1857			*																		
Eucyclops serrulatus Fischer, 1851																					
Mesocyclops sp.																	*				
Paracyclops fimbriatus Fischer, 1853																					
Tropocyclops prasinus Fischer, 1860		*		*		*	*	*							*						
Cyclopoid nauplii			*	*	*	*	*		*		*	*	*		*	*	*	*	*	*	*
Harpacticoid Copepod																					
Attheyella lewisae Wells, 2007								*	*	*	*	*	*	*							
Harpacticoid nauplii	*																				
Tardigrades			*				*				*		*						*		
Oligochaetes					*	*		*			*		*			*	*				
Ostracods	*		*		*		*	*		*	*		*				*	*			
Chronomids			*	*		*	*					*	*						*		
Springtails		*	*	*	*	*	*	*	*		*	*	*	*		*	*	*	*	*	
Mites		*	*		*		*	*	*		*	*				*		*		*	
Gastrotrichs																					

Table 3.12 Microfauna recorded at all sites during December 2011

Species	N1	N2	N3	N4	N5	N6	N7	L1	L5	L6	L7	D2	D3	D4	D5	D6	
Rotifera																	
Bdelloids																	
Aspelta angusta Harring & Myers, 1928													*				
Cupelopagis vorax Leidy, 1857																	
Lecane bulla Gosse, 1851																	
L. closterocerca Schmarda, 1859																	
L. hamata Stokes, 1896																	
L. lunaris Ehrenbrg, 1832																	
L. pusilla Harring, 1914																	
Notommata allantois Wulfert, 1935																	
Polyarthra dolichoptera Idelson 1925																	
Proales decipiens Ehrenberg, 1832					*												
Scaridium longicaudum Müller, 1786									*								
Squatinella mutica Ehrenberg, 1832																	
Tetrasiphon hydrocora Ehrenberg, 1840													*				
Trichocerca similis Wierzejski, 1893																	
T. tigris Müller, 1786																	
Trichocera sp.													*				
Trichotria tetractis Ehrenberg, 1830																	
Cladocera																	
Alona guttata Sars, 1862																	
A. quadrangularis Müller, 1776																	
Camptocercus australis Sars, 1896																	
Ceriodaphnia dubia Richard, 1894																	
Chydorus sp. Müller, 1785					*	*	*	*	*			*	*		*	*	
Ilyocryptus sordidus Lièvin, 1848																	
Oxyurella tenuicaudis Sars, 1862																	
Simocephalus vetulus Müller, 1776							*						*			*	
Copepoda																	
Cyclopoid copepod	*	*	*	*	*	*	*	*	*	*		*	*	*	*	*	
Acanthocyclops robustus Sars, 1863	*			*	*	*	*						*		*	*	
Diacyclops bicuspidatus Claus, 1857																	
Eucyclops serrulatus Fischer, 1851										*				*			
Mesocyclops sp.																	
Paracyclops fimbriatus Fischer, 1853																	
Tropocyclops prasinus Fischer, 1860		*	*				*		*			*					
Cyclopoid nauplii	*		*	*	*	*	*	*	*	*		*	*		*	*	
Harpacticoid Copepod																	
Attheyella lewisae Wells, 2007																	
Harpacticoid nauplii							*										
Tardigrades						*						*					
Oligochaetes																*	
Ostracods	*	*	*	*	*		*	*		*	*	*	*	*	*	*	*
Chronomids						*	*		*		*	*	*			*	*
Springtails												*				*	*
Mites				*		*						*	*	*		*	
Gastrotrichs																	

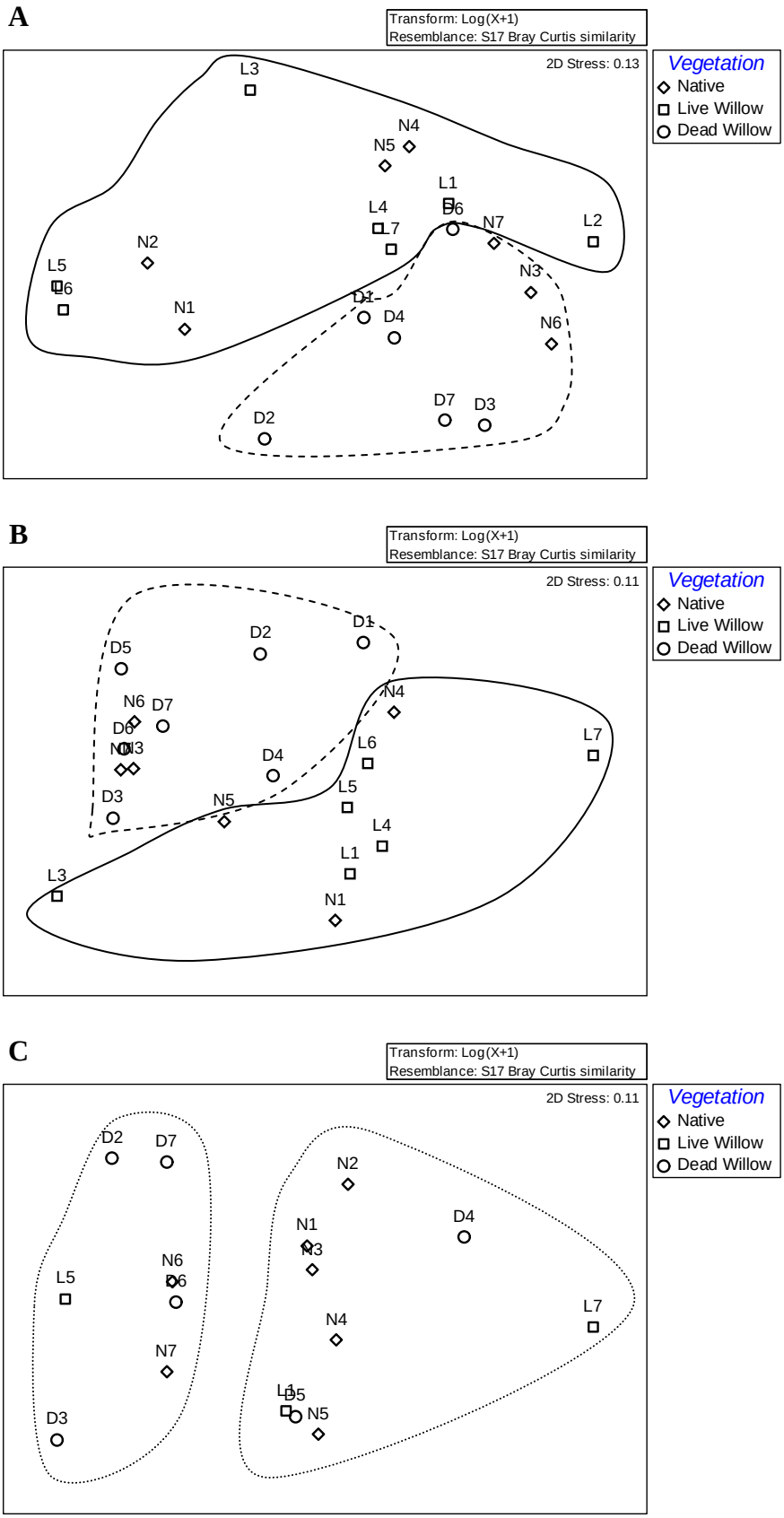


Figure 3.13 Multi-dimensional scaling (MDS) plots showing community composition and species distribution for A) February, B) July and C) December
Live Willow cluster - - - - - Dead Willow Cluster --------- Combination Cluster .....

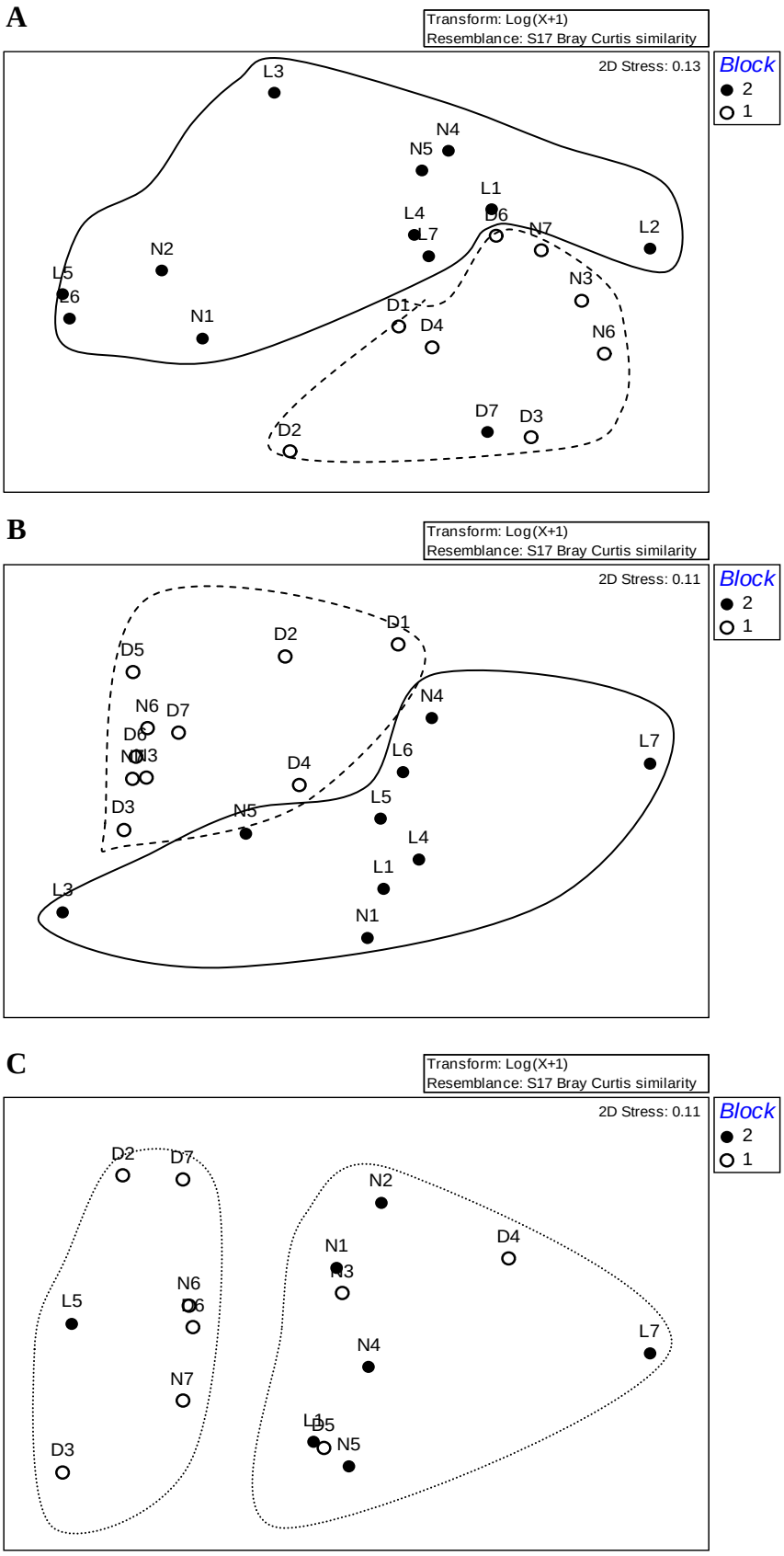


Figure 3.14 Multi-dimensional scaling (MDS) plots showing community composition and species distribution for Blocks 1 & 2 during A) February, B) July and C) December
Block 1 - - - - - Block 2 ------- Combination Cluster .....

Table 3.13 One-Way ANOSIM of native, live willow and dead willow of microfaunal community composition and species distribution for February, July and December

	February	July	December
P - Value	0.474	0.116	0.272
		R-Statistic	
Native/Live Willow	-0.096	-0.034	0.272
Native/Dead Willow	0.054	-0.046	0.053
Live Willow/Dead Willow	0.089	0.336	-0.080

Table 3.14 One-Way ANOSIM for the influence of the sand bar on microfaunal community composition and species distribution for February, July and December

	February P Value	July P Value	December P Value
Sand Bar	0.024	0.012	0.388

3.2 Discussion

3.2.1 Environmental variables

Artificially controlled lake levels did not coincide with seasonal rainfall patterns during my study period. Lake levels peaked in late summer of 2011 during a period of low recorded rainfall. However, Eser (1998) found that from February 1995- November 1997, the artificial control of lake level fluctuations corresponded with monthly rainfall, where in summer months rainfall and lake levels were low and winter months rainfall and lake levels were high. Block 2 of the Waiotaka Scenic Reserve, located closest to the lake shore, demonstrated the highest recorded average water depth of 19.4 cm in February, which coincided with high lake levels. When water levels were at highest in Block 2 in my study, water level in Block 1 was 14.7 cm. However, a higher water level was recorded in Block 1 in July, with a water depth of 16.2 cm, compared to 12.3 cm in Block 2. During this time lake levels had consistently declined and rainfall had rapidly increased, suggesting that Block 1 water levels are related to high rainfall. Similarly, in December Block 1 had a higher water depth of 15.4 cm compared to 9.0 cm in Block 2, where lake levels had declined and rainfall peaked before declining in early summer. Overall, recorded wetland water depths, along with patterns in lake level and seasonal rainfall, suggest that the wetland reserve blocks, separated by the sandbar, could possibly be influenced by different hydrological factors. My results are supported by Eser (1998), who demonstrated that surface water levels in the Stump Bay wetland were correlated with rainfall, season and lake levels in areas of close proximity to the lake shore.

Lowest water temperatures were recorded in winter ranging from 6.5-7.3˚C and were highest in early summer ranging from 16.7-18.1˚C and late summer ranging from19.2-19.7˚C, these results are expected based on New Zealand seasonal temperate cycles. Values of pH remained in a narrow range between 6.3-6.7 throughout the study, and are typical measurements for swamp wetlands (Peters & Clarkson 2010), with low levels in early and late summer and higher levels in winter. Specific conductance for early and late summer ranged from high values of 129.2-209.7 µS/cm and 129.4 µS/cm in winter. High values of specific conductance in early and late summer and low levels in winter may be influenced

by rainfall input into the wetland, where low conductance occurs during time of heavy rainfall (Winterbourn & McDiffett 1996). There were no significant differences of physiochemical variables amongst native, live willow and dead willow, with the exception of dissolved oxygen in February, and canopy density in all seasons. Dissolved oxygen concentrations ranged from 0.2-0.6 mg/L in early summer, 0.2-1.1 mg/L in late summer, and 0.5-1.8 mg/L in winter, with levels around dead willows consistently higher, lower among live willows and natives with concentrations in between. ANOVA indicated a significant difference in dissolved oxygen between the live and dead willow in February, suggesting that the warmer summer temperatures (average 19.2˚C) and the decomposition of fallen leaves from the live willow may have been a contributing factor in the low dissolved oxygen found under live willow (Read & Barmuta 1999). Low dissolved oxygen during winter in Block 2 with the presence of live willow could be due to willows dropping their leaves. ANOVA showed no significant difference in dissolved oxygen between natives and the live or dead willows, this is likely due to natives being distributed within both of the two blocks. The consistent low dissolved oxygen in Block 2, and relatively high dissolved oxygen concentrations in Block 1, could also be explained by hydrological variations in Blocks 1 and 2. ANOVA analysis for canopy density indicated significant differences between live and dead willow in all seasons, where live willow canopy density much higher than dead willow canopy density. This was expected for February and December, as *Salix cinerea* is in full bloom from early to late summer (Webb et al. 1988). This does not explain canopy density between live and dead willow in July, however, as live willow lose its leaves in winter; it was thus expected that there would be no significant difference between live and dead willow. The dead willow trees had been poisoned three years prior to my study and they had started to break down with branches breaking away from the main trunk and decomposing, which could account for similar canopy densities in July. With high canopy density cover among willow during the summer period we might also have expected to see lower chlorophyll *a* concentrations and water temperature due to shading. For example Glover & Saga (1994) and Lester et al.'s (1994) studies on small rivers and streams demonstrated that willows shade out algal production.

Overall, apart from shading and dissolved oxygen levels, environmental conditions of live and dead willow stands in this study seemingly made no significant difference to environmental variables, relative to natives. This could be due to the willow trees representing stand-alone individuals, with a continuous canopy not yet formed. The density of willows seems to play a major factor affecting ecological impact on streams and rivers (Collier 1994; Glover & Sagar 1994; Johnston 2011; Lester et al. 1994), which has been observed in this wetland study.

Ground cover in native, live willow and dead willow sites consisted of diverse vegetation types and abundance. Native sites were primarily dominated by *Baumea rubiginosa* vegetation with a mix of *Carex secta, Typha orientalis* (raupo) and *Cortaderia toetoe* (toetoe) and open water. Ground cover in live willow sites consisted of a combination of *B. rubiginosa* and open water and small mix of *C. secta*, toetoe, *Leptocarpus similis* (oioi) , *Coprosma robusta* (karamu) and raupo. Ground cover associated with dead willows was mostly open water with *B. rubiginosa* and a mixture of with oioi, karamu, *Phormium tenax* (harakeke) and young shoots of *Salix cinerea*. The vegetation found in my study is typical of wetland ground cover previously recorded in the Waiotaka Scenic Reserve (Department of Conservation 2002). The ground cover of both live and dead willow sites were made up of similar vegetation found in Eser's (1998) study of vegetation communities within the South Taupo Wetland. The main vegetation classes invaded by *S. cinerea* were *Baumea* sedgelands, oioi rushlands, raupo reedlands and open water communities, suggesting that *S. cinerea* has the capability to invade the Waiotaka Scenic Reserve and potentially establish dense canopies. The ground cover of native sites was predominantly *B. rubiginosa*, however as *S. cinerea* established the ground cover of *B. rubiginosa* was reduced dramatically and open water was increased, suggesting that *S. cinerea* altered the original vegetation ground cover. Young shoots of *S. cinerea* found among dead willow demonstrates that *S. cinerea* is capable of resprouting following control treatment which is supported by Ray and Davenhill's (1996) trials of ground control methods. Ray and Davenhill trialled the drill and inject method and found that *S. cinerea* were prone to resprouting from the roots.

3.2.2 Microfaunal Composition and Dynamics

The abundant taxa found in the study were copepods, including cyclopoid copepods such as *Acanthocyclops robustus*, *Diacyclops bicuspidatus*, *Eucyclops serralatus*, *Mesocyclops* sp., *Tropocyclops prisinus*, cyclopoid nauplii, the harpacticoid copepod *Attheyella lewisae*, cladocerans including *Chydorus* sp., *Ceriodaphnia dubia* and *Simocephalus vetulus*, and ostracods. Although there were high numbers of copepods and cladocerans, there was a high diversity of rotifer species found in my study. Species found consisted of bdelloids, *Aspelta angusta*, *Cupelopagis vorax, Lecane bulla, L. closterocerca, L. hamata, L. lunaris, L. pusilla*, *Notomatta allantois*, *Polyarthra dolichoptera*, *Proales decipiens*, *Scaridium longicaudum, Tetrasiphon hydrocora Trichocerca similis*, and *T. tigris*.

ANOVA indicated that there were no significant differences in microfaunal species richness between native, live and dead willow sites in any season. However, the MDS ordination and ANOSIM results of species composition indicated that microfaunal assemblages were clustered in groups either side of the sand bar, suggesting that Blocks 1 and 2 functioned independently. This may be influenced by hydrological differences between Block 1 and 2 of the wetland reserve with the presence of fluctuating lake levels and seasonal rainfall, as mentioned in earlier discussion, throughout the wetland. These three main taxa of copepods, cladocerans and ostracods are common wetland inhabitants found in various wetland worldwide and their presence are determined by diverse and complex factors typical to the nature of wetland systems such as hydrologic fluctuations (Mitsch & Gosselink 2007) and turbidity (Kirk & Gilbert 1990). The influence of hydrology on the structure and function of wetlands can impact on the biological diversity and productivity of wetlands. Hydroperiod plays a factor influencing the diversity and structure of temporary wetland microfaunal assemblages. Waterkyn et al. (2008), for example, found a positive relationship between species richness and hydroperiod in temporary wetlands of Tour du Valat, Rhône delta, France. The study found that temporary wetlands supported a rich diversity of cladocerans and cyclopoid copepods, suggesting that temporary wetlands provide rich food resources and reduced stress from biotic factors. In wetlands with longer hydroperiods, more time is available for completion of life cycles, colonisation and community development (Schneider & Frost 1996).

Variable water depths throughout the wetland could contribute to the major type of species found in the study. Zimmer et al. (2001) found in prairie wetlands, United States, that *S. vetulus*, cyclopid copepods and ostracods occupied increasing depth of the wetland water column.

No significant differences were found in microfaunal species composition and richness between native, live and dead willows. This result is similar to Suren and Sorrell's (2010) spatial study of aquatic invertebrates in four lowland wetlands of the South Island, New Zealand, which found that invertebrate communities varied between different wetlands rather than between habitats or plants within a wetland. Their study found that invertebrate community composition and percentage abundance were relatively similar between areas with or without vegetation. Kratzer and Batzer (2007) also found very little variation in invertebrate communities in Okefenokee Swamp, Florida, USA, despite sampling in different plant habitats of swamplands. This was attributed to the fact that water quality did not vary greatly throughout the wetland, and therefore supported similar invertebrate communities, despite different habitats. In contrast, Ryan (2001) found that microfaunal assemblages in peat wetland of the Whangamarino wetland, New Zealand corresponded closely with vegetative regions where dense willow sites were distinct from natives, demonstrating a clear separation of vegetation classes based on microfaunal species.

My study found higher abundance of copepods than cladocerans in each season, along with a high diversity of rotifer species. Chittapun et al. (2009) studied the diversity and composition of zooplankton in rice fields of Panthum Thani province, Thailand. The study found a high diversity of rotifer taxa with a low diversity of copepods and cladocerans. Chittapun et al. found that rotifers reached much higher densities when cladoceran abundance was low. Cladoceran abundance decreased as copepod abundance increased suggesting that the presence of copepods reduces the growth rate of cladocerans. This was further supported by Chittapun et al.'s study which demonstrated that cladocerans and copepods appear to play a role in the structure of rotifer diversity.

Shiel and Green (1996) provided a complete list of all rotifer species recorded in New Zealand to that time. Most of the rotifer species found in my study are listed by Shiel and Green (1996) except for *Tetrasiphon hydracora*. Serafim et al.'s (2003) study of rotifers in the Upper Paraná River Floodplain found *T. hydracora* in the littoral zone of the floodplain. *Aspelta angusta, L. bulla*, *S. longicaudum*, and *T. tigris* were also found with and *T. hydracora* in the littoral zone, typical of wetland inhabitants.

3.3 References

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CHAPTER FOUR: SHORT-TERM EFFECTS OF MICROFAUNAL COMMUNITY COMPOSITION POST WILLOW TREATMENT

4.1 Results

4.1.1 Environmental Variables

In February, water depth (Figure 4.1A) in Block 2 averaged 15.0 cm for treatment and 18.0 cm under control trees, and rose slightly to 17.2 cm for control and 19.3 cm treatment in March (Figure 4.1B). T-tests indicated there was no significant difference of water depth between control and treatment before, and control and treatment after ($P>0.05$; Table 4.1). Lake Taupo water levels (Figure 4.2) declined over summer along with low rainfall (Figure 4.3). The average water temperature in February (Figure 4.4A) ranged from 16.8˚C for treatment and 17.5 ˚C for control, and declined slightly to 13.9 ˚C for control and treatment in March (Figure 4.4B). T-test analyses showed no significant differences of water temperature between control and treatment before, and control and treatment after ($P>0.05$; Table 4.2). The average pH (Figure 4.5A) was 6.7 for treatment and 6.8 for control in February, and decreased to 6.3 for control and 6.4 for treatment in March (Figure 4.5B). T-tests indicated no significant difference of pH between control before and treatment before, or control after and treatment after ($P>0.05$; Table 4.3). The average dissolved oxygen was 0.3 mg/L for control and treatment in February (Figure 4.6A), and increased slightly to 0.4 mg/L for treatment and 0.8 mg/L for control in March (Figure 4.6B). However, t-tests indicated no significant difference in dissolved oxygen concentrations between control and treatment before, and control and treatment after ($P>0.05$; Table 4.4). Average specific conductance was 210.3 µS/cm for treatment and 294.4 µS/cm for control in February (Figure 4.7A), and decreased remarkably to 98.2 µS/cm for treatment and 102.6 µS/cm for control in March (Figure 4.7B). T-tests indicated a significant difference of specific conductance between control before and treatment before (P value=0.025; Table 4.5), but did not show a significant difference for control after and treatment after (P value=0.608; Table 4.5). The average chlorophyll a for February (Figure 4.8A) was low, ranging from 0.09 µg/L for control and 0.29 µg/L for treatment, and increased slightly to 1.4 µg/L

for control and 1.9 µg/L treatment in March (Figure 4.8B). T-test results indicated there was no significant difference of chlorophyll a between control before and treatment before, and control after and treatment after (P>0.05; Table 4.6). Average canopy density ranged from 66.7% for treatment and 72.8 % for control in February (Figure 4.9A), and canopy density for March (Figure 4.9B) was 79.4 % for control while treatment declined to 46.1 %. T-test analysis indicated there was a significant difference between control and treatment after (P value=0.005; Table 4.7) Control and treatment after trees are illustrated in figures 4.10 and 4.11. Treatment trees were poisoned with a metsulfuron herbicide mix. The amount of metsulfuron herbicide mix injected into the individuals was dependant on the height and stump diameter of the tree (the stump diameter was not measured in this study). The tallest tree at 3.5 m was injected with the most herbicide mix of 11.6 g/mL, trees of 3 m in height were injected with 4.0-6.3 g/mL of herbicide mix, and the shortest at 2.5 m was injected 9.0 g/mL of herbicide mix (Table 4.8).

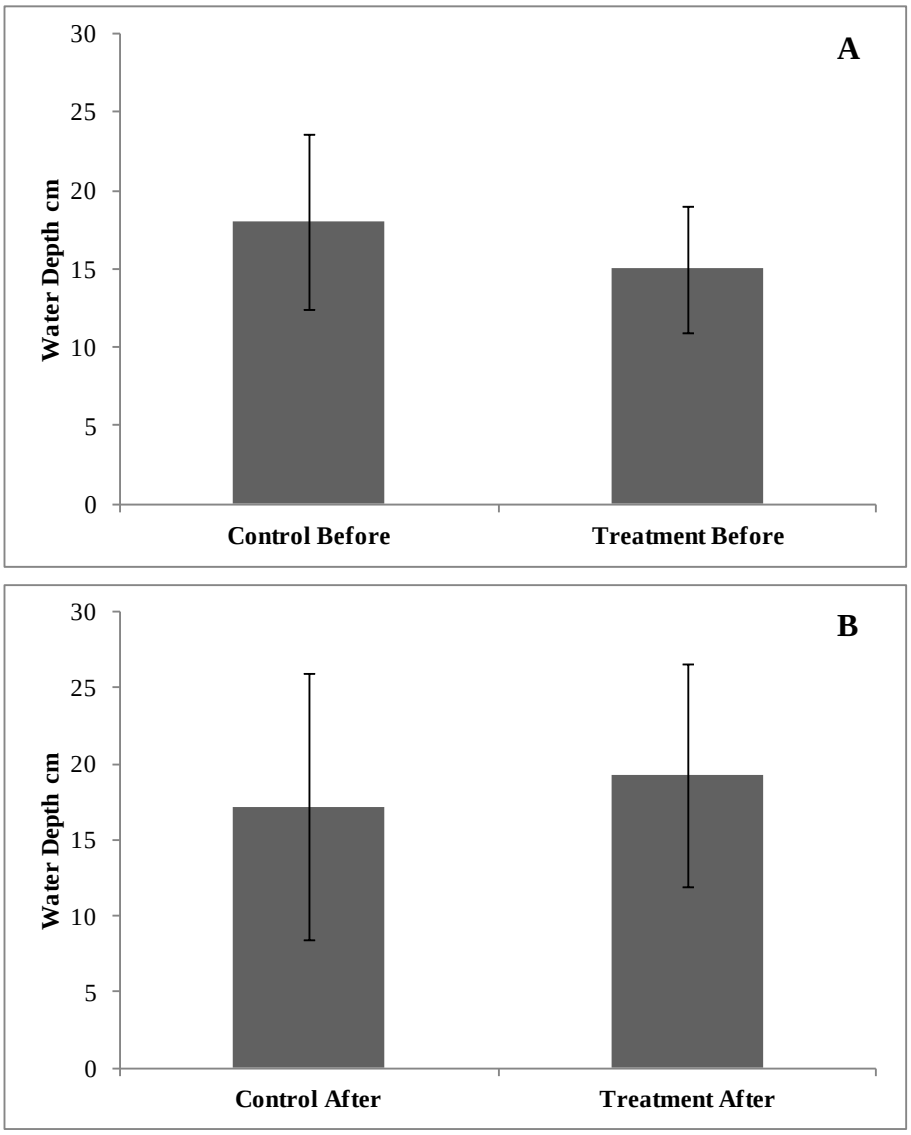

Figure 4.1 Mean±SD water depth of A) before and B) after treatment

Table 4.1 Summary of T-Test results for mean water depth for before and after treatment showing t-values and probability values (P)

Water Depth cm	t-value	P
Control Before/Treatment Before	0.722	0.497
Control After/Treatment After	-0.539	0.604

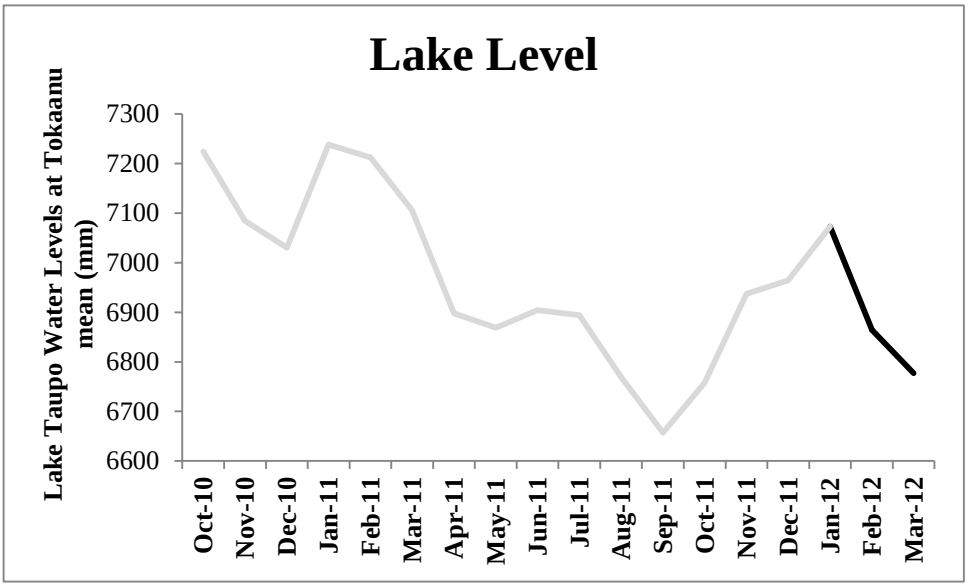

Figure 4.2 Lake Taupo at Tokaanu mean monthly lake levels from October 2010 – March 2012. Sample period between February – March 2010
Data from NIWA, Tokaanu

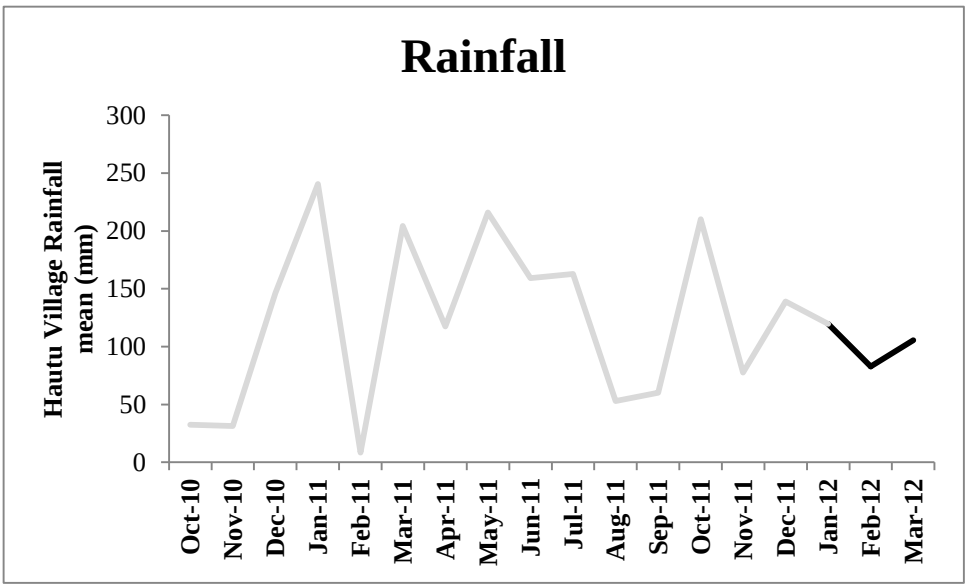

Figure 4.3 Hautu Village Station mean monthly rainfall from October 2010 – March 2012. Sample period February – March 2012
Data from http://cliflo.niwa.co.nz/pls/niwp/doc/terms.html

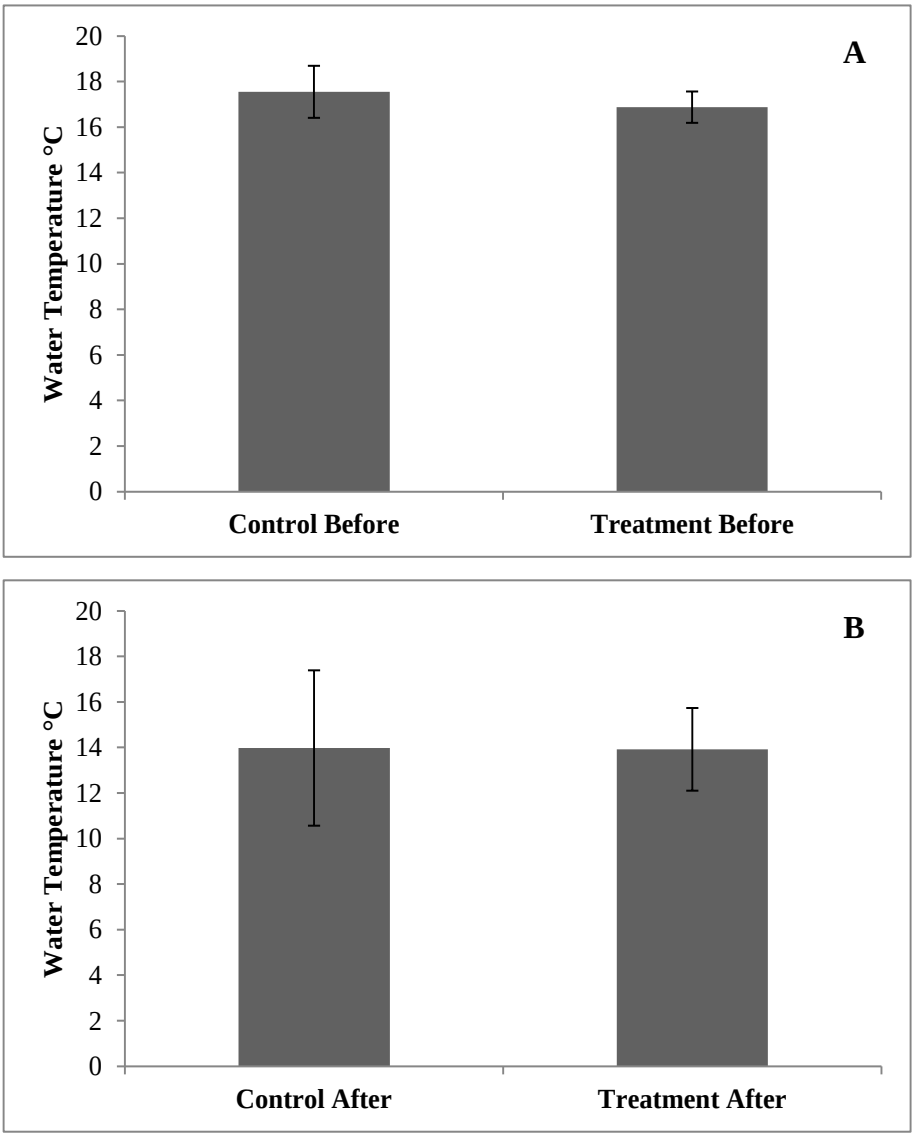

Figure 4.4 Mean±SD water temperature of A) before and B) after treatment

Table 4.2 Summary of T-Test results for mean water temperature for before and after treatment showing t-values and probability values (P)

Water Temperature °C	t-value	P
Control Before/Treatment Before	0.996	0.357
Control After/Treatment After	-0.098	0.923

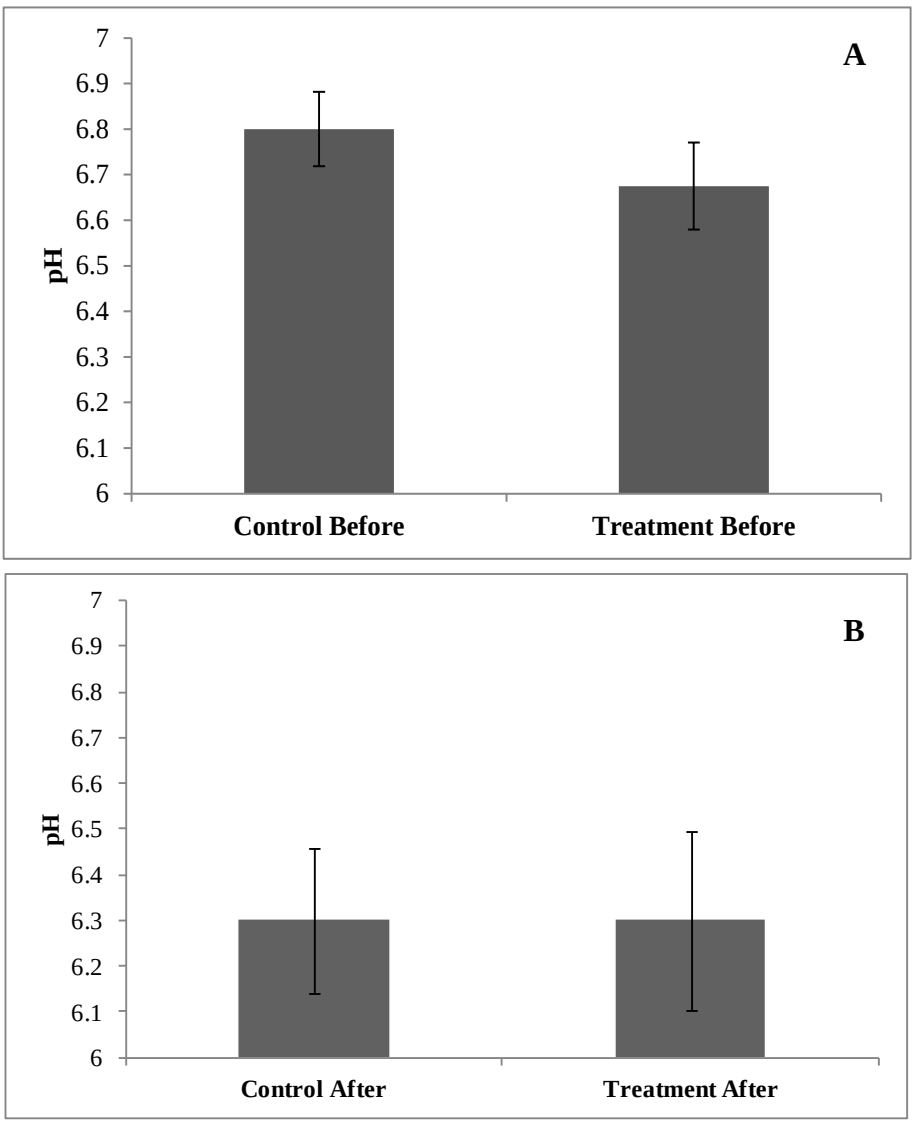

Figure 4.5 Mean±SD pH of A) before and B) after treatment

Table 4.3 Summary of T-Test results for mean pH for before and after treatment showing t-values, and probability values (P)

pH	t-value	P
Control Before/Treatment Before	1.989	0.093
Control After/Treatment After	-0.351	0.734

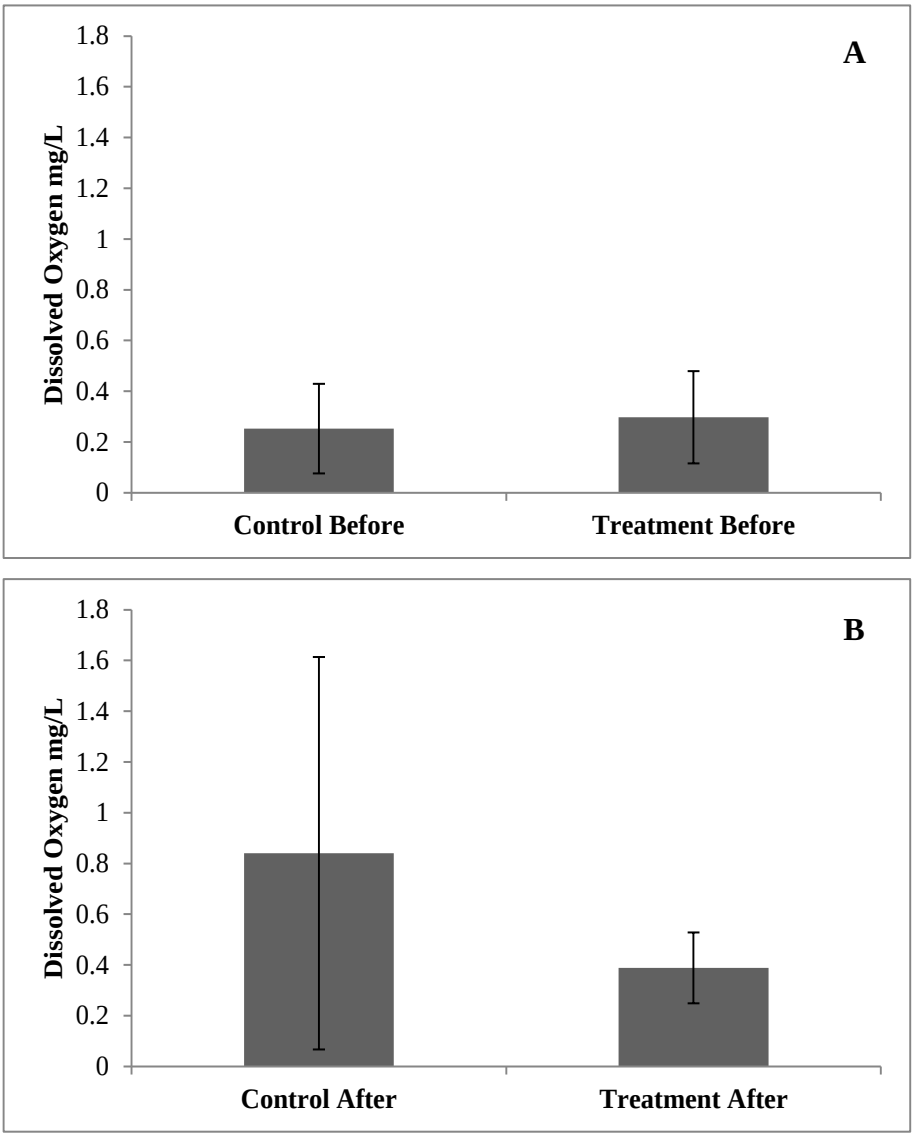

Figure 4.6 Mean±SD dissolved oxygen of A) before and B) after treatment

Table 4.4 Summary of T-Test results for mean dissolved oxygen for before and after treatment showing sum t-values and probability values (P)

Dissolved Oxygen mg/L	t-value	P
Control Before/Treatment Before	-0.635	0.548
Control After/Treatment After	0.224	0.827

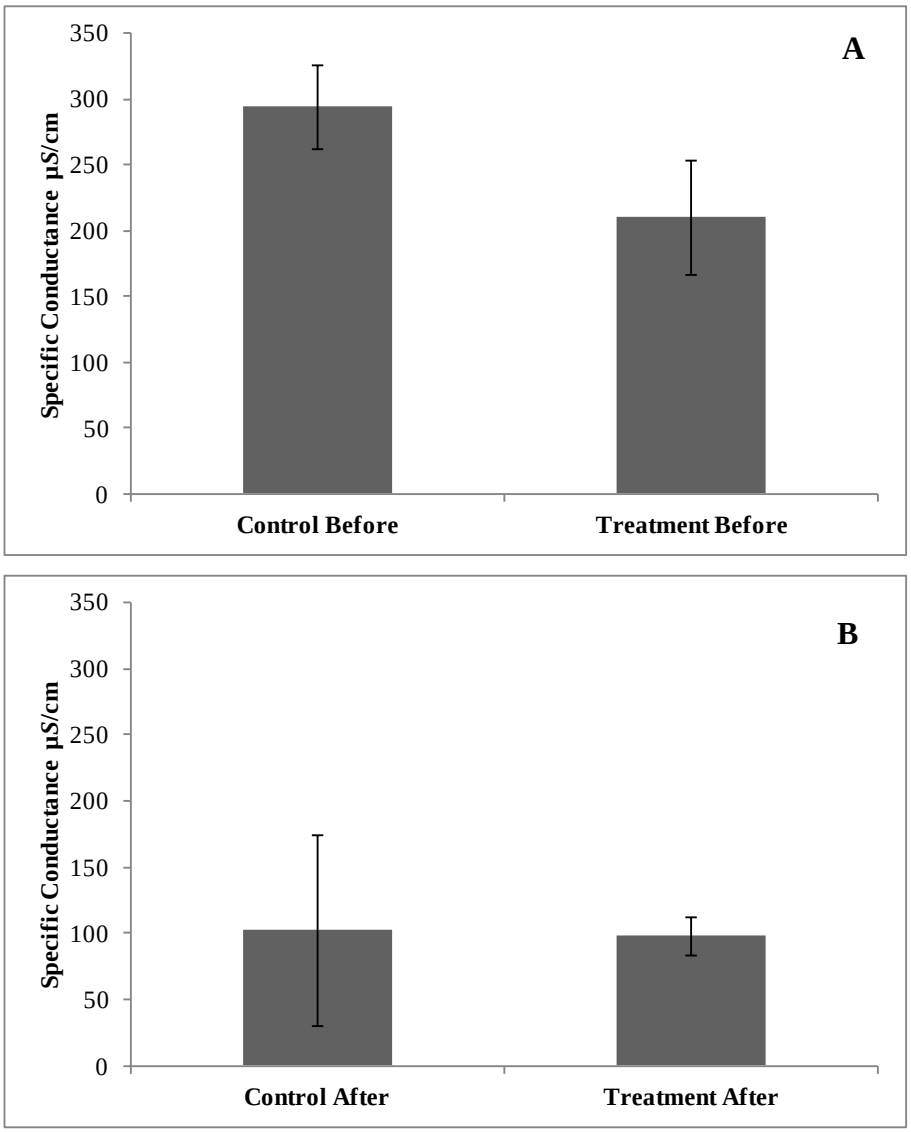

Figure 4.7 Mean±SD specific conductance of A) before and B) after treatment

Table 4.5 Summary of T-Test results for mean specific conductance for before and after treatment showing t-values and probability values (P)

Specific Conductance µS/cm	t-value	P
Control Before/Treatment Before	2.950	0.025
Control After/Treatment After	-0.533	0.608

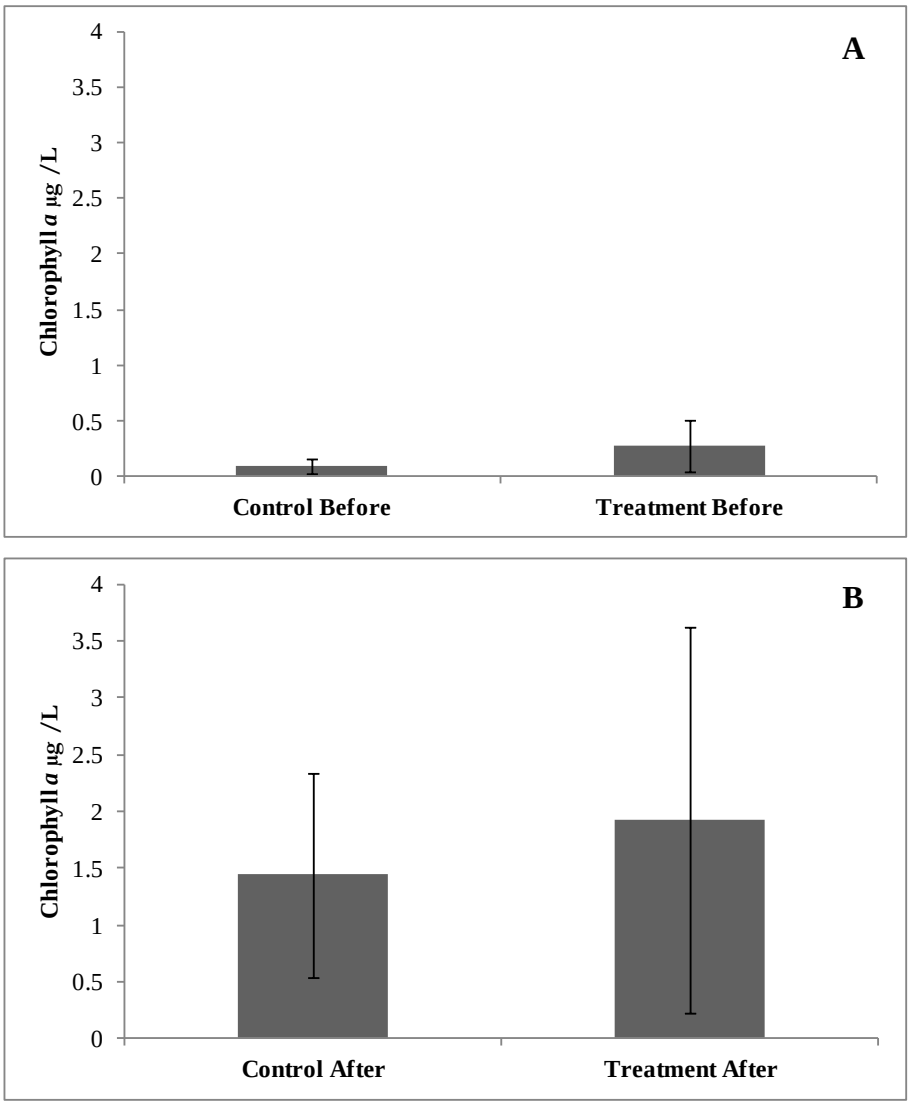

Figure 4.8 Mean±SD chlorophyll *a* of A) before and B) after treatment

Table 4.6 Summary of T-Test results for mean chlorophyll *a* for before and after treatment showing t-values and probability values (P)

Chlorophyll *a* µg/L	t-value	P
Control Before/Treatment Before	0.336	0.747
Control After/Treatment After	0.065	0.949

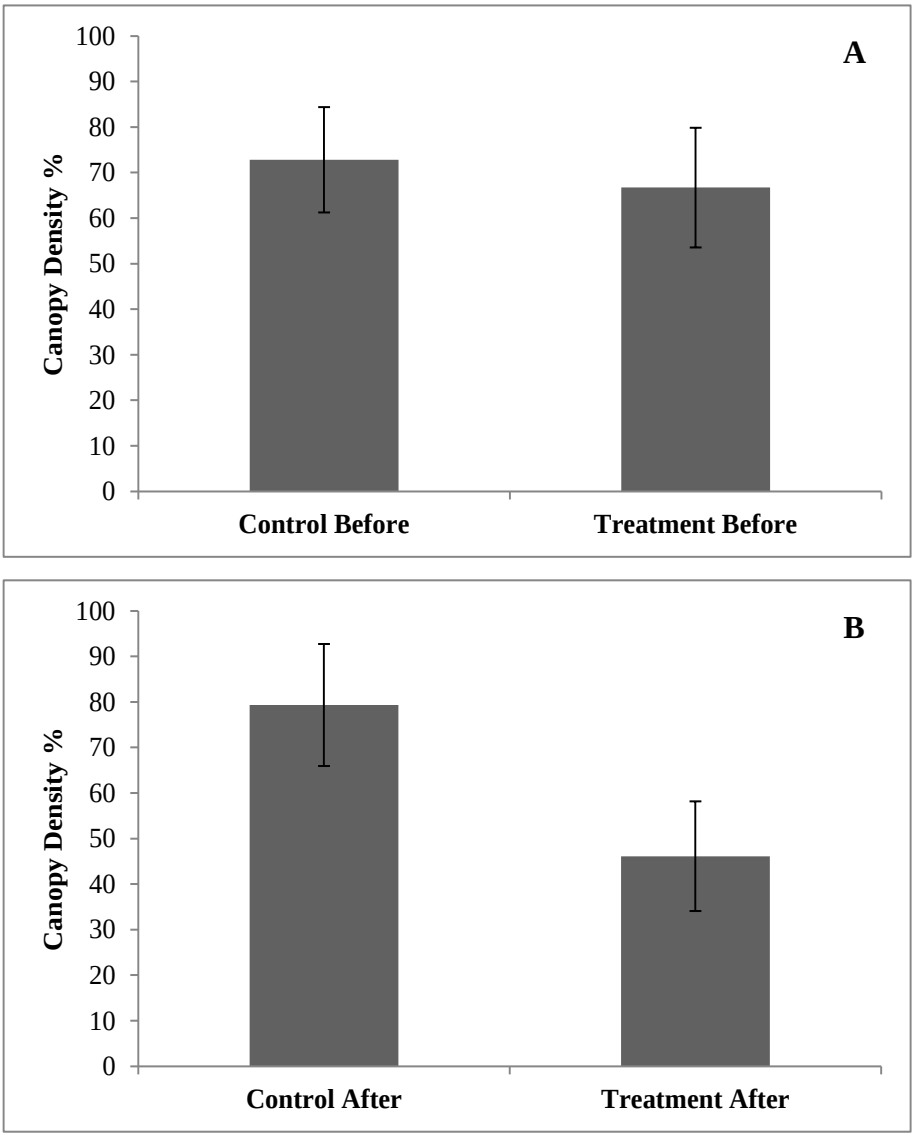

Figure 4.9 Mean±SD canopy density of A) before and B) after treatment

Table 4.7 Summary of T-Test results for mean canopy density for before and after treatment showing t-values and probability values (P)

Canopy Density %	t-value	P
Control Before/Treatment Before	0.747	0.482
Control After/Treatment After	3.770	0.005

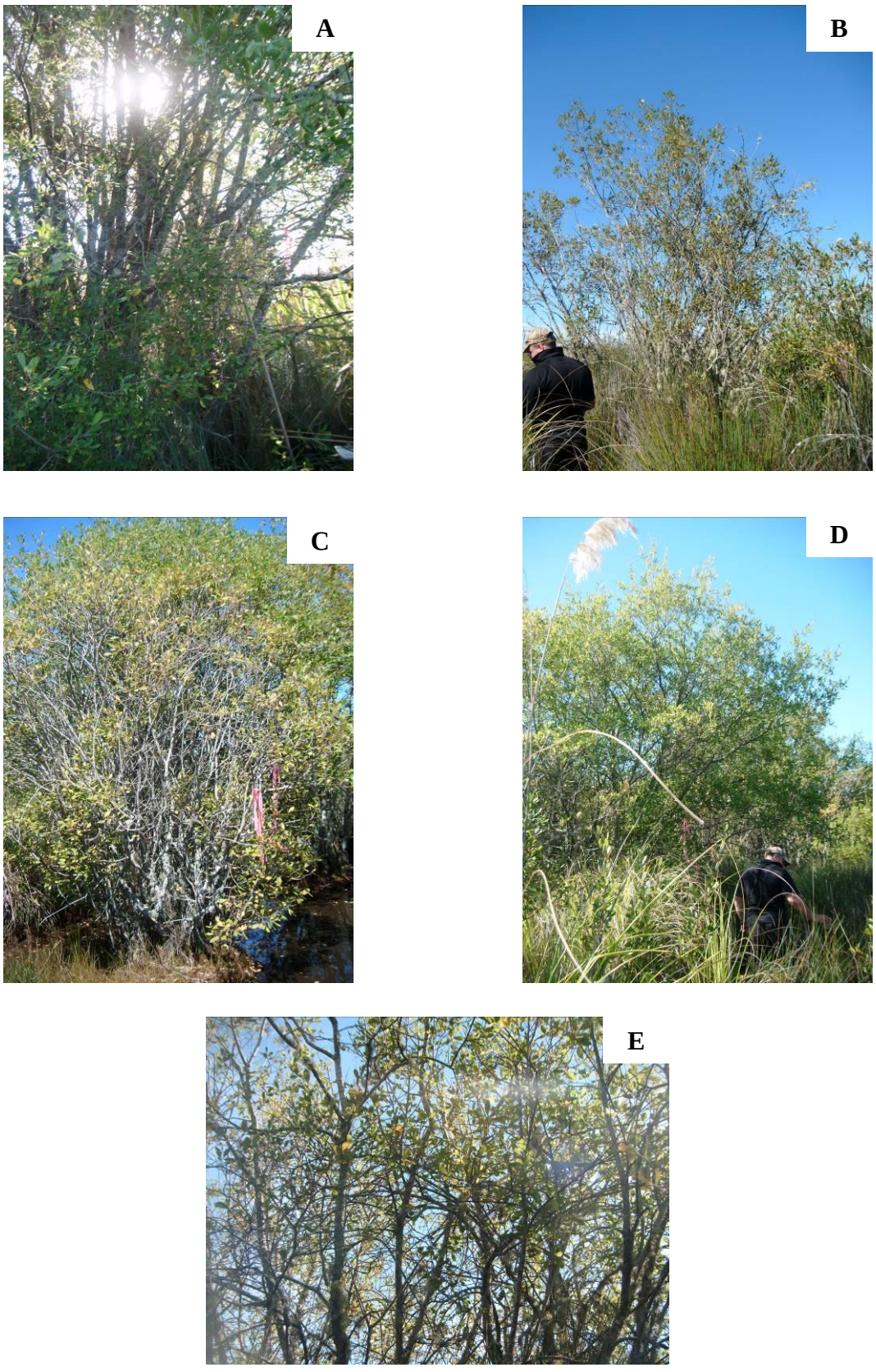

Figure 4.10 Control After A) L3, B) L5, C) L6, D) L8 & E) L13

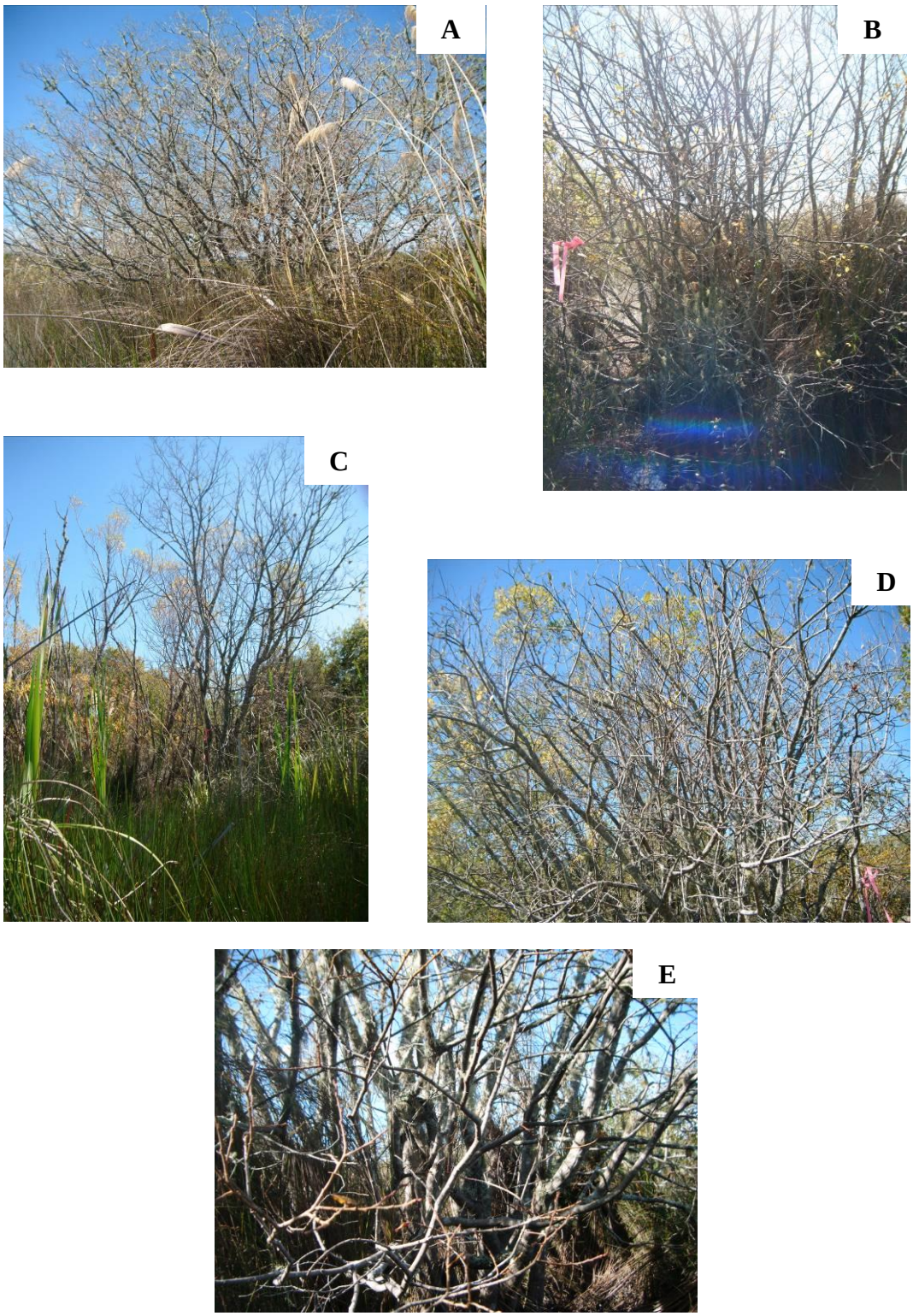

Figure 4.11 Treatment After A) P1, B) P4, C) P3, D) P2 & E) P5

Table 4.8 Metsulfuron chemical mix g/mL injected into each treated tree

	P1	P2	P3	P4	P5
Height of tree m	3.5	3	3	3	2.5
Metsulfuron g/mL	11.6	5	4	6.2	9

4.1.2 Microfaunal Composition and Dynamics

Species richness was slightly higher in the before samples in February 2012 (Figure 4.12A) with an average of five species found in control and treatment. However species abundance was much lower than that found in February 2011, with an average of 38 copepods, nine cladocerans and two ostracods in 2011 compared to eight copepods, one cladoceran and five ostracods. The major taxa recorded (Figure 4.13A) consisted of copepods, including cyclopoid copepods such as *Eucyclops serrulatus*, *Tropocyclops prasinus*, cyclopoid nauplii and harpacticoid nauplii and cladocerans included *Chydorus* sp. and *Simocephalus vetulus,* and ostracods. Rotifers were also observed in low abundance. These included bdelloid rotifers and *Scaridium longicardum* (Table 4.10). In March (after) species richness for the control sites averaged five species and treatment sites averaged three species (Figure 4.12B). The major taxa recorded (Figure 4.13B) consisted of copepods, including the cyclopoid copepods *E. serrulatus, Paracyclops fimbriatus, T. prasinus,* cyclopoid nauplii and the harpacticoid copepod *Attheyella lewisae*, cladocerans such as *Chydorus* sp. and *S. vetulus,* and ostracods (Table 4.10). Copepods increased in March with six copepods in live willow and 14 copepods in treated willow. Bdelloid rotifers were also observed in low abundance among live willow.

The MDS ordination for control and treatment before (Figure 4.14) did not present any clear clusters to indicate any significant differences between microfaunal species composition. ANOSIM results for control and treatment before did not indicate a significant difference between control and treatment ($P>0.05$; Table 4.11). Also the MDS ordination for control and treatment after (Figure 4.15) indicated no significant differences between microfaunal species composition. ANOSIM results for control and treatment after revealed no significance between control and treatment ($P>0.05$; Table 4.11).

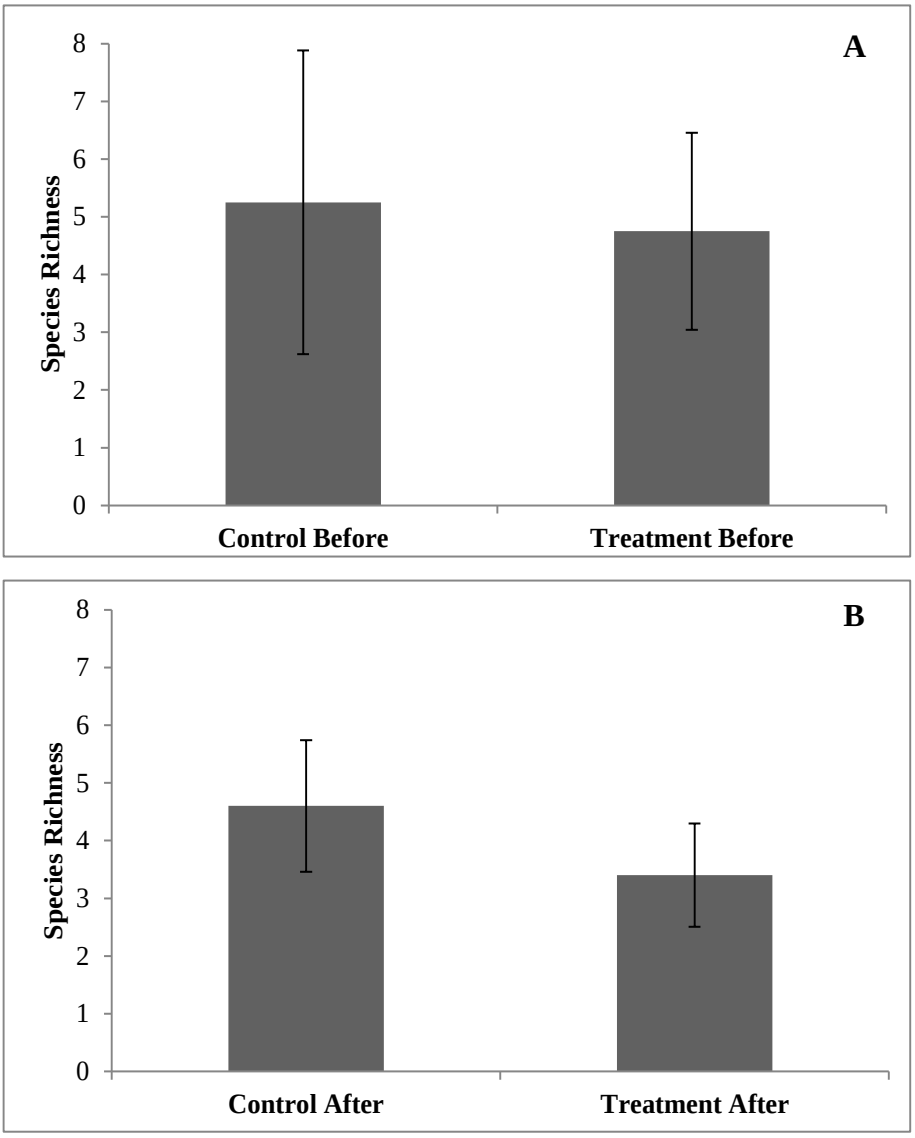

Figure 4.12 Mean±SD species richness of A) before and B) after treatment

Table 4.9 Summary of T-Test result for mean species richness for before and after treatment showing t-values and probability values (P)

Species Richness	t-value	P
Control Before/Treatment Before	0.142	0.891
Control After/Treatment After	1.708	0.125

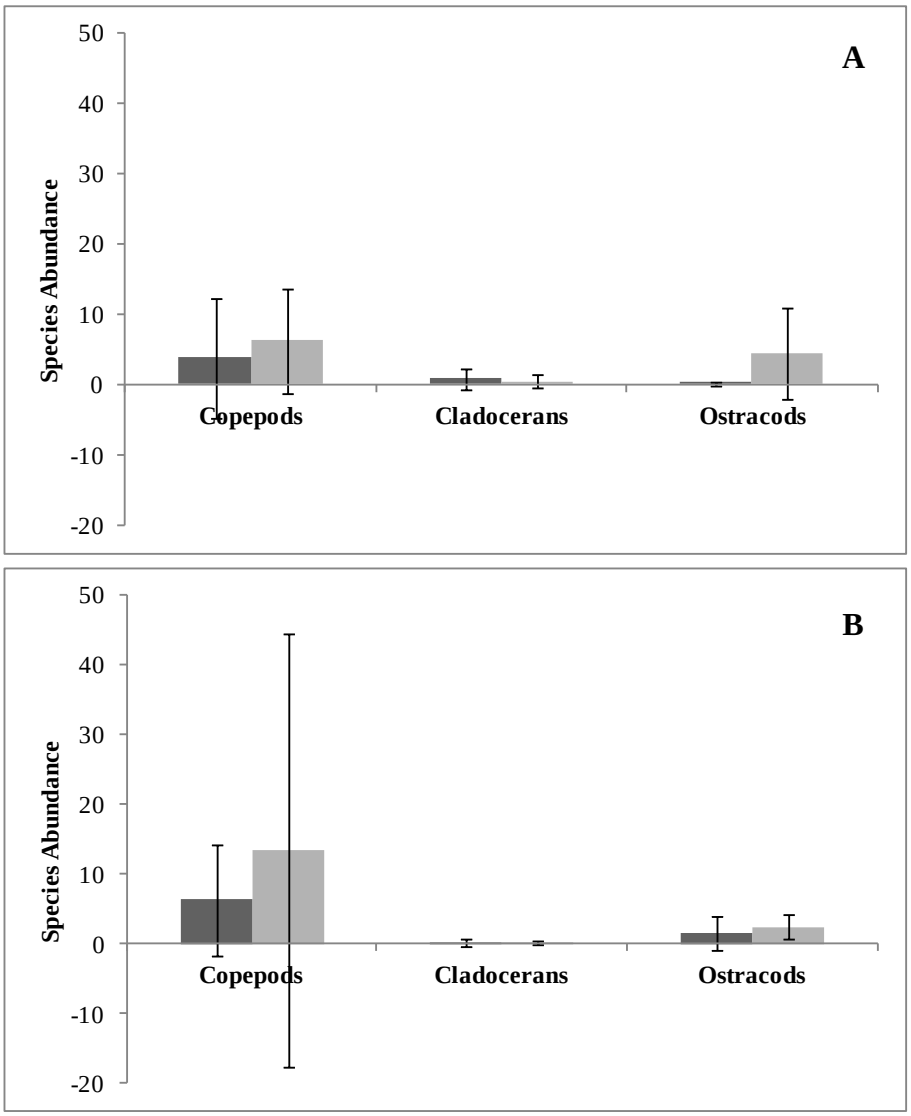

Figure 4.13 Mean±SD species abundance for A) before and B) after treatment

Table 4.10 Microfauna recorded at all sites during before and after treatment

Species	L5	L6	L8	L13	L9	L10	L11	L12	L3	L5	L6	L8	L13	P1	P2	P3	P4	P5
Rotifera																		
Bdelloids	*			*				*					*					
Aspelta angusta Harring & Myers, 1928																		
Cupelopagis vorax Leidy, 1857																		
Lecane bulla Gosse, 1851																		
L. closterocerca Schmarda, 1859																		
L. hamata Stokes, 1896																		
L. lunaris Ehrenbrg, 1832																		
L. pusilla Harring, 1914																		
Notommata allantois Wulfert, 1935																		
Polyarthra dolichoptera Idelson 1925																		
Proales decipiens Ehrenberg, 1832																		
Scaridium longicaudum Müller, 1786	*			*		*												
Squatinella mutica Ehrenberg, 1832																		
Tetrasiphon hydrocora Ehrenberg, 1840																		
Trichocerca similis Wierzejski, 1893																		
T. tigris Müller, 1786																		
Trichocera sp.																		
Trichotria tetractis Ehrenberg, 1830																		
Cladocera																		
Alona guttata Sars, 1862																		
A. quadrangularis Müller, 1776																		
Camptocercus australis Sars, 1896																		
Ceriodaphnia dubia Richard, 1894																		
Chydorus sp. Müller, 1785	*			*		*		*					*	*		*		
Ilyocryptus sordidus Lièvin, 1848																		
Oxyurella tenuicaudis Sars, 1862																		
Simocephalus vetulus Müller, 1776	*			*		*							*					
Copepoda																		
Cyclopoid copepod	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*	*
Acanthocyclops robustus Sars, 1863																		
Diacyclops bicuspidatus Claus, 1857																		
Eucyclops serrulatus Fischer, 1851							*		*			*				*		*
Mesocyclops sp.																		
Paracyclops fimbriatus Fischer, 1853										*								
Tropocyclops prasinus Fischer, 1860	*		*			*		*	*	*		*		*		*		*
Cyclopoid nauplii	*	*	*	*	*	*		*	*	*	*	*	*	*		*		*
Harpacticoid Copepod																		
Attheyella lewisae Wells, 2007									*									
Harpacticoid nauplii			*				*		*		*	*			*		*	*
Tardigrades	*				*													
Oligochaetes																		
Ostracods				*	*	*	*	*	*	*	*	*	*	*		*	*	*
Chronomids																		
Springtails																		
Mites	*	*				*					*	*						
Gastrotrichs																		

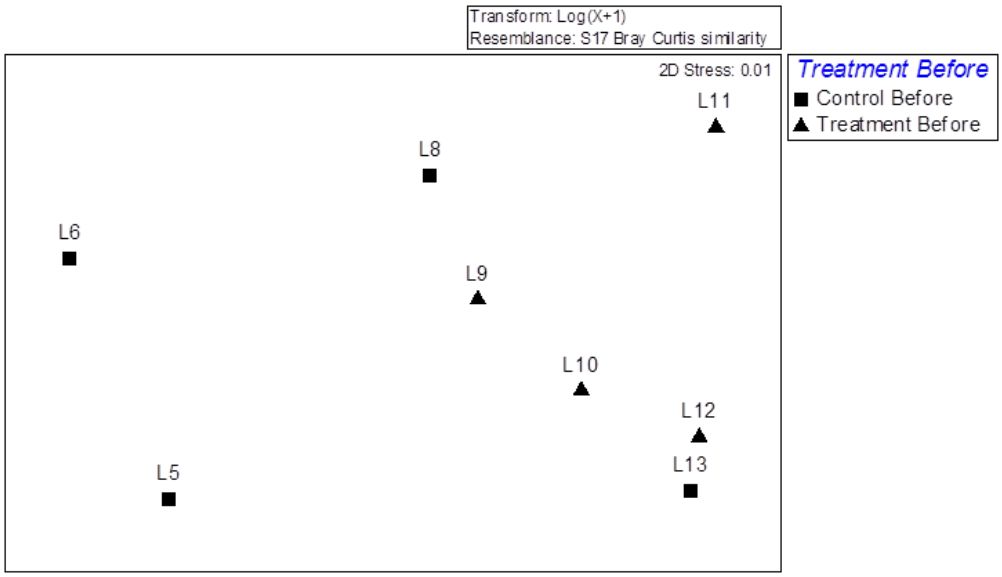

Figure 4.14 Multi-dimensional scaling (MDS) plots showing microfaunal community composition and species distribution for before treatment

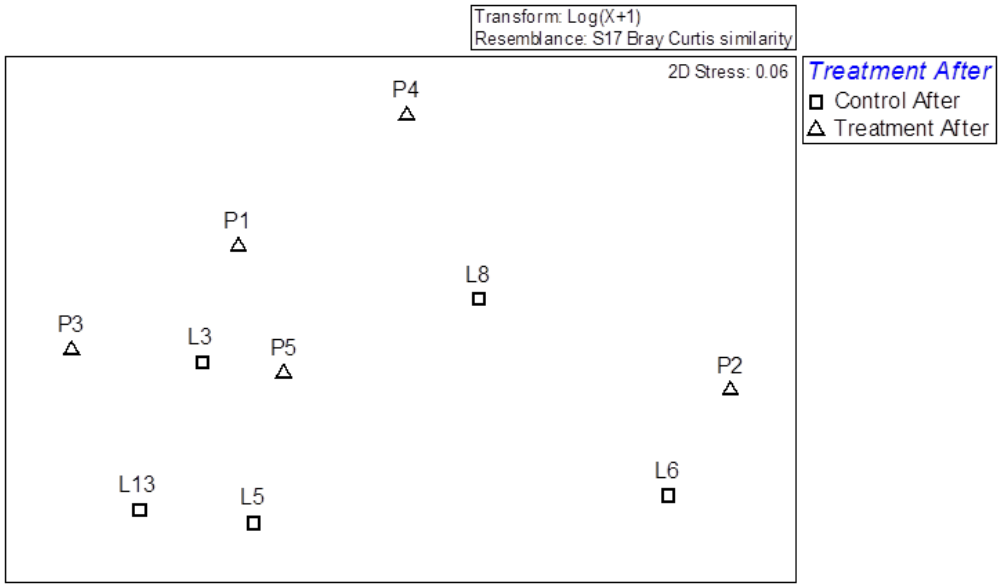

Figure 4.15 Multi-dimensional scaling (MDS) plots showing microfaunal community composition and species distribution for after treatment

Table 4.11 ANOSIM One-Way Analysis for the influence of the sand bar on microfaunal community composition and species distribution for before and after control and treatment

February	P Value	R Value	March	P Value	R Value
Control Before/ Treatment Before	0.257	0.188	Control After/ Treatment After	0.683	0.072

4.2 Discussion

4.2.1 Environmental Variables

My results indicated that artificial control of lake levels do not coincide with seasonal rainfall events. Lake levels were lower in February of 2012 compared to the high levels observed in February 2011. Similarly rainfall levels were higher in February 2012 compared to low levels reported in summer of 2011. In February of 2011 the average water depth for Block 2 of the Waiotaka Wetland Reserve was recorded at 19.4 cm, while water depth in February 2012 averaged from 15-18 cm. The 2012 results is a slight decrease from the previous year which may be due to the changes in artificial control of lake levels and seasonal rainfall. In March, average water depth rose slightly ranging from 17.2-19.3 cm. My results are similar to those of Eser (1998) who demonstrated surface water levels in the Stump Bay wetland were influenced by lake levels in areas of close proximity to the lake shore.

My results recorded for water temperature in late summer 2012 ranged from 13.9-17.6˚C, which is similar to late summer 2011 water temperature average of 19.2˚C in February and 13.9˚C in March. Values of pH remained in a narrow range between 6.3-6.8 during February and March which is typical of swamp wetlands (Peters & Clarkson 2010). Dissolved oxygen concentrations averaged 0.3 mg/L in February 2012, similar to the dissolved oxygen levels of 0.2 mg/L among live willow in February 2011. Dissolved oxygen increased slightly in March with treated willows low at 0.4 mg/L and control willows at 0.8 mg/L. Specific conductance for February 2012 ranged from 210.3-294.4 µS/cm, which was within range to the reported 214.7 µS/cm in February 2011. High values of specific conductance in late summer could be due to low rainfall experienced over summer. Chlorophyll *a* was low in February ranging from 0.1-0.3 µg/L, possibly due to the high canopy density ranging from 66.7-72.8%. In March chlorophyll *a* increased slightly ranging from 1.4 µg/L for control willows and 1.9 µg/L for treated willows. This could have been be due to leaf fall of the treated trees with a canopy density of 46.1% compared to control willows that remained at a high density canopy cover of 79.4%. This is further demonstrated in t-test analysis indicating a significant difference between the control and treatment willows (P

value=0.005). The decomposition of fallen leaves after treatment may have been a contributing factor to lower dissolved oxygen levels and higher chlorophyll *a* concentrations found under treated willow (Read & Barmuta 1999). Specific conductance for February 2012 ranged from 210.3-294.4 μS/cm which was remarkably different to the recorded 214.7 μS/cm in February 2011. High values of specific conductance in late summer 2012 could be due to low rainfall experienced over summer. T-test analysis indicated a significant difference of specific conductance between control and treatment before (P value=0.025), however, while they may be statistically different it is unlikely to be ecologically significant as all of the live willows in February were sampled in Block 2 on the same day.

There were no significant differences of physiochemical variables between control and treatment before, nor control and treatment after, with the exception of specific conductance in February and percentage canopy cover. It was expected that there would be no differences between control before and treatment before, as they essentially were all live willows sampled within the same block of the Waiotaka Wetland reserve on the same day. However, we did expect to see significant differences between control after and treatment after as treated individuals were poisoned and dropped leaves. Differences such as water temperature, dissolved oxygen, pH and chlorophyll *a* should have been remarkably different between control and after treatment due to the application of metsulfuron herbicide, opening of the canopy and added detritus to the aquatic system.

The study found no significant differences between physiochemical variables between the control and treatment after the application of herbicide mix metsulfuron. Thompson et. al. (2006) explains that herbicides are absorbed by plants, soils and sediments and rapidly degrades, therefore limiting the potential for significant indirect inputs to surface waters. This is further supported by Tatum (2004) who clarifies that herbicides such as metsulfuron-methyl is soluble in water and rapidly degrades once it enters the environment. Tatum also states that metsulfuron-methyl does not persist or bioaccumulate in water as these herbicides are designed to act specifically on vegetation. Golombieski et al. (2008)

reiterates that 12 hours after metsulfuron-methyl was applied it was not detected in the water.

4.2.2 Microfaunal Composition and Dynamics

The taxa found in the study were copepods, including cyclopoid copepods such as *Eucyclops serralatus*, *Paracyclops fimbriatus*, *Tropocyclops prasinus*, cyclopoid nauplii, the harpacticoid copepod *Attheyella lewisae,* cladocerans including *Chydorus* sp. and *Simocephalus vetulus,* and ostracods. Rotifera included bdelloids and *Scaridium longicaudum,* which were only found among live willow.

According to MDS ordination for control and treatment before there were no clear clusters between species composition, this was supported by ANOSIM results of $P>0.05$. This was to be expected as all sample sites were live willow within the same block on the same day. Also the MDS ordination for control and treatment after poisoning resulted with no significant differences between microfaunal species and composition. However we did expect to see significant differences due to the poisoning and killing of trees resulting in the addition of metsulfuron herbicide, the opening of the canopy and leaf fall into the aquatic system.

Ground control application of metsulfuron was used for this study and found no significant differences between microfaunal species and composition. Golombieski et al. (2008) studied the effects of the herbicide metsulfuron-methyl on cladocerans, copepods and rotifers in rice farms of Rio Grande do Sul State, Brazil. The study found that the application of metsulfuron-methyl did not affect microfaunal communities, as the herbicide was not detected in the water 12 hours after application. Fowlkes et al.'s (2003) study of the imazapyr herbicide and its effects on benthic macroinvertebrates in a cypress wetland, United States. The study found no significant differences of taxa richness and abundance observed among control or treatment blocks. Gardner and Grue (1996) studied the aerial application effects of the systemic herbicide Garlon® on aquatic invertebrate species in two wetlands, in central Washington, United States. The study found no significant differences in the abundance of invertebrates collected before and after treatment, suggesting that Garlon® does not pose a threat to aquatic invertebrates

in wetlands, as Garlon® did not persist in toxic concentrations. This was supported by Kreutzweiser et al.'s (1989) investigation of stream invertebrate response to aerial application of Roundup® and found that movements of most invertebrates were not affected.

The opening of the canopy and leaf fall into the water column following willow treatment found no significant differences between microfaunal species and composition. However, there was a slight increase of copepod density among treated willows than that found among the live willows. This may be explained by Quinn et al.'s (1997) study of willow shade and the association of benthic invertebrate taxon richness, in Mangaotama stream, New Zealand. The study found invertebrate densities declined with canopy density of 60-90%. Therefore, the opening of the canopy to 40% may have assisted with increased copepod density. Also, the increase of leaf litter may have played a role in the slight increase of copepod density. Studies have demonstrated that willow leaves are palatable to biota in streams (Lester et al. 1994). Lester et al. (1994) found that willow leaves were broadly used both directly and indirectly as a food source by organisms of all functional feeding groups in willow lined reaches of two Central Otago streams in New Zealand. Although these studies were specifically examining benthic invertebrates in streams, this does explain the change in canopy density and leaf litter deposited into the water column post willow treatment observed in my study.

4.3 References

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CHAPTER FIVE: CONCLUSION

The aim of this research was to quantitatively examine microfaunal assemblage abundance, richness and community composition among *Salix cinerea* stands within freshwater wetlands and determine whether these microfaunal assemblages are affected by willow growth and willow control treatment.

5.1 Long-term effects of microfaunal community composition between native vegetation versus live and dead *Salix cinerea*

The long-term effects of microfaunal community composition were examined among native, live and dead *S. cinerea*. Natives represented indigenous wetland plant species and also reflected the most favourable communities for *S. cinerea* to potentially invade. These sites consisted of raupo (*Typha orientalis*), toetoe (*Cortaderia toetoe*) and sedges including *Baumea rubiginosa* and *Carex secta*, and open water. Sampling was undertaken in February (late summer), July (winter) and December (early summer) 2011, to encompass seasonal variation. During these times *S. cinerea* was in late summer bloom, had lost their leaves (winter), or were in early summer bloom, respectively.

The study found no significant differences of environmental variables amongst native, live and dead *S. cinerea*, with the exception of dissolved oxygen in February, and canopy density in all seasons. ANOVA indicated a significant difference in dissolved oxygen between the live and dead *S. cinerea* in February, suggesting that the warmer summer temperatures (average 19.2˚C), combined with the decomposition of fallen leaves from the live *S. cinerea*, may have been a contributing factor in the low dissolved oxygen found under live willow. ANOVA analysis for canopy density indicated significant differences between live and dead *S. cinerea* in all seasons, where live *S. cinerea* canopy density was higher than dead *S. cinerea* canopy density. This was expected for February and December, as *Salix cinerea* is in full bloom from early to late summer. Overall, apart from shading and dissolved oxygen levels, environmental conditions of live and dead *S. cinerea* stands in this study seemingly made no significant difference

to environmental variables, relative to natives. This is possibly due to the *S. cinerea* trees representing stand-alone individuals, with a continuous canopy not yet formed.

The abundant taxa found in the study were copepods, including cyclopoid copepods such as *Acanthocyclops robustus*, *Diacyclops bicuspidatus*, *Eucyclops serralatus*, *Mesocyclops* sp., *Tropocyclops prasinus*, cyclopoid nauplii, the harpacticoid copepod *Attheyella lewisae*, cladocerans including *Chydorus* sp., *Ceriodaphnia dubia* and *Simocephalus vetulus*, and ostracods. Although there were high numbers of copepods and cladocerans, there was a high diversity of rotifer species found in my study. Species found consisted of bdelloids, *Aspelta angusta*, *Cupelopagis vorax, Lecane bulla, L. closterocerca, L. hamata, L. lunaris, L. pusilla*, *Notomatta allantois*, *Polyarthra dolichoptera*, *Proales decipiens*, *Scaridium longicaudum*, *Trichocerca similis*, and *T. tigris*, and included *Tetrasiphon hydrocora*, which is recorded for the first time in New Zealand.

ANOVA indicated that there were no significant differences in microfaunal species richness between native, live and dead *S. cinerea* in any season. However, the MDS ordination and ANOSIM results of species composition indicated that microfaunal assemblages were clustered in groups either side of the sand bar, suggesting that Blocks 1 and 2 functioned independently. This may be influenced by hydrological differences between Block 1 and 2 of the wetland reserve, with differing responses to fluctuating lake levels and seasonal rainfall, suggesting that microfaunal communities are regulated by hydrology rather than by the presence of willows or willow control.

5.2 Short-term effects of microfaunal community composition post willow control treatment

The short-term effects of microfaunal community composition were examined post willow control treatment. My study found no significant differences in environmental variables between control and treatment before, nor control and treatment after, with the exception of specific conductance in February and

percentage canopy cover. Treated *S. cinerea* trees died and lost their leaves after ground application of metsulfuron.

It was expected that there would be no differences between control and treatment before, as they essentially were all live willows sampled within the same block of the Waiotaka Wetland reserve on the same day. However, we did expect to see significant differences between control and treatment after as treated individuals were poisoned, killed and dropped their leaves. Differences such as water temperature, dissolved oxygen, pH and chlorophyll *a* might have been expected to be remarkably different between control and after treatment due to the application of metsulfuron herbicide, the opening of the canopy and added detritus to the aquatic system. The results of the study was supported by claims that herbicides such as metsulfuron are absorbed by plants, soils and sediments and rapidly degrades, therefore limiting the potential for significant indirect inputs to surface waters.

The taxa found in this experiment were copepods, including cyclopoid copepods such as *Eucyclops serralatus*, *Paracyclops fimbriatus*, *Tropocyclops prasinus*, cyclopoid nauplii, the harpacticoid copepod *Attheyella lewisae*, cladocerans including *Chydorus* sp. and *Simocephalus vetulus*, and ostracods. Rotifera included bdelloids and *Scaridium longicaudum*, which were only found among live willow.

According to the MDS ordination for control and treatment before there were no clear clusters between species composition. This was supported by non-significant ANOSIM results. This was to be expected as all sample sites were live willow within the same block on the same day. Also, the MDS ordination for control and treatment after poisoning resulted in no significant differences between microfaunal species and composition. However we did expect to see significant differences due to the poisoning and killing of trees resulting in the application of metsulfuron herbicide, the opening of the canopy and leaf fall to the aquatic system.

The results of the study was supported by observations that the application of herbicide does not seem to affect microfaunal communities, as the herbicide was not detected in the water 12 hours after application. Ground control application seemed to have no significant impact on microfaunal assemblages post treatment. This was also seen in studies of aerial application of herbicides where there were no significant differences in the abundance of invertebrates collected before and after treatment, suggesting that the herbicide does not pose a threat to aquatic invertebrates in wetlands as it did not persist in toxic concentrations.

Overall, the findings of this study indicate that the presence of *S. cinerea* seemed to make no significant difference to microfaunal abundance and diversity, possibly due to stand alone individuals rather than the formation of a dense canopy. Furthermore, ground control treatment of *S. cinerea* using metsulfuron had no direct or indirect impacts to microfaunal abundance and diversity. However, had the study been undertaken under a dense canopy of *S. cinerea* it is likely that the results may potentially be different.