The effects of macrophyte control on freshwater fish communities and water quality in New Zealand streams

Michael John Crawshaw Greer

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Frontispiece

A view of Craws Creek; a heavily modified waterway in the catchment of Waituna Lagoon that contains a healthy population of giant kokopu (*Galaxias argenteus*).

Abstract

The ecological effects of macrophyte removal on New Zealand's stream ecosystems are not well understood. To address this issue, I investigated how native fish abundance and diversity are impacted by mechanical excavation of macrophytes, and examined the role suspended sediment and dissolved oxygen play in driving changes in community structure following macrophyte removal. Population surveys conducted before and after experimental mechanical excavation of macrophytes demonstrated that native fish abundance was reduced by 52 percent after excavation, but species diversity was not affected. Although partial macrophyte removal was still found to reduce fish abundance significantly, radiotelemetry of giant kokopu (*Galaxias argenteus*) demonstrated that this technique might prevent large individuals of this species from leaving targeted waterways.

Suspended sediment was monitored before and after a large macrophyte removal operation (> 80 kilometres of waterway excavated) by water sampling and continual turbidity measurements. Immediate and dramatic increases in suspended sediment concentrations were observed during and immediately after mechanical excavation of macrophytes (120,000 % increase). Suspended sediment concentrations remained elevated for 77 days after macrophyte removal, and were particularly high during periods of flooding. Sediment concentrations regularly exceeded concentrations required to elicit an avoidance response in juvenile migratory native fish and introduced salmonids, suggesting fish abundance may be reduced. Respirometry trials demonstrated that suspended sediment concentrations recorded after macrophyte removal had no effect on the respiratory performance of brown trout (*Salmo trutta*). However, laboratory-based feeding experiments demonstrated that the same sediment concentrations can reduce the feeding rates of this species by up to 43 percent. In addition, suspended sediment can affect other aspects of water quality to the extent that fish may be affected.

Dissolved oxygen concentrations were continually monitored in several streams following mechanical excavation of macrophytes and herbicide application. Statistically detectable reductions in dissolved oxygen concentration, associated with the resuspension of highly organic anoxic sediment, were observed immediately after mechanical excavation of macrophytes. Time spent in moderate (the daily percentage of DO measurements below 30 percent saturation) and severe hypoxia (the daily percentage of DO measurements below 30 percent saturation) was 43 percent and 37 percent greater in the three-day period after mechanical excavation than in the three-day period before. At a number of sites included in this study hypoxia persisted for a several

days, and previous research suggests this would have been sufficient to cause significant mortality in the resident fish population. Periods of hypoxia associated with plant decomposition were also observed following herbicide application. However, reductions in dissolved oxygen were gradual, and it is expected that fish were able to move out of treated waterways before oxygen conditions became lethal.

The results of this study indicate that mechanical macrophyte removal is likely to cause significant adverse impacts on fish communities in New Zealand waterways. Furthermore, increased sediment resuspension and associated changes in dissolved oxygen concentration after macrophyte removal may have a greater impact on resident fish than previously thought.

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Chapter 1.0: General introduction

1.1 Disturbance

The influence of disturbance regime on community structure is recognised as an important component of the ecology of freshwater environments, especially streams (Sousa, 1984; Resh et al., 1988; Reice et al., 1990; Melo et al., 2003). White and Pickett (1985) defined disturbance as any event that "disrupts ecosystem, community or population structure and changes resources, substrate availability or the physical environment". The ecological effects of disturbances are dependent on a number of factors, including the frequency, intensity and spatial extent of disturbance and the surviving faunal assemblage (Resh et al., 1988; Reice et al., 1990). It is predicted that communities in infrequently disturbed environments reflect the outcomes of inter-specific interactions, and the species present resist competition and predation, but are susceptible to environmental changes during disturbance (Connell, 1978; Menge and Sutherland, 1987; Petraitis et al., 1989; Reice et al., 1990; Werner and Anholt, 1993; Wootton et al., 1996). Conversely, communities in frequently disturbed systems are expected to comprise species with a high resistance/resilience to disturbance, but a low resistance to competition and predation in stable environments (Connell, 1978; Menge and Sutherland, 1987; Petraitis et al., 1989; Reice et al., 1990; Werner and Anholt, 1993; Closs and Lake, 1996; Wootton et al., 1996).

Credited as the founder of the intermediate disturbance hypothesis (Menge and Sutherland, 1987; Resh et al., 1988; Reice et al., 1990), Connell (1978) predicted that species diversity is greatest in systems where disturbance is neither too frequent nor infrequent. In this situation species with a low resistance/resilience to disturbance are able to persist, and the availability of disturbed areas allows for continual colonisation by resistant/resilient species that would otherwise be outcompeted or predated upon (Connell, 1978). The occurrence of frequent and severe natural disturbances, such as floods and droughts, can reduce species diversity (Fausch and Bramblett, 1991; Lake, 2003; Melo et al., 2003; Davey and Kelly, 2007), but the disturbances that pose the biggest threat to freshwater ecosystems are caused by humans (Resh et al., 1988; Maitland, 1995; Dudgeon et al., 2006).

Anthropogenic activities alter ecosystem function (Resh et al., 1988; Niemi et al., 1990; Reice et al., 1990; Maitland, 1995; Ward, 1998), and are recognised as major contributors to the current global decline of riverine biodiversity (Maitland, 1995; Groombridge and Jenkins, 1998; Ward, 1998; Ricciardi and Rasmussen, 1999; Dudgeon et al., 2006). The relative importance of different anthropogenic disturbances, such as pollution, water abstraction, modified flow regimes, habitat removal and sedimentation, varies between regions depending on land use practices (Resh et al., 1988; Sala et al., 2000; Thieme et al., 2005; Dudgeon et al., 2006). Agriculture, logging, industry,

urbanisation, construction, hydroelectric generation and water impoundment all play key roles in disturbing freshwater ecosystems (Resh et al., 1988; Maitland, 1995; Groombridge and Jenkins, 1998; Thieme et al., 2005; Dudgeon et al., 2006). Although there is a widespread understanding that such activities have universally reduced the abundance and distribution of freshwater native fish (Westlake et al., 1972; Maitland, 1995; Groombridge and Jenkins, 1998; Ricciardi and Rasmussen, 1999; Sala et al., 2000), the effects of many forms of anthropogenic disturbance on fishes are yet to be quantified (Maitland, 1995). Fish are an important component of aquatic ecosystems, and changes in their abundance can have a disproportionately large effect on benthic community structure (Maitland, 1995). Therefore, effective management of lotic ecosystems relies on a thorough understanding of all anthropogenic threats to fish.

The effects of anthropogenic disturbances on fish communities in drainage networks are not well understood, primarily because the ecological value of these systems has been ignored in the past (Herzon and Helenius, 2008). Drains increase connectivity within landscapes and provide important habitat for aquatic fauna (Herzon and Helenius, 2008; Colvin et al., 2009). The taxonomic richness of invertebrates and fish in drains is often comparable to nearby natural streams (Armitage et al., 2003), and in some regions they contain a more diverse faunal assemblage than unmodified waterways (Simon and Travis, 2011). Drains are also used as refuges by fish and invertebrates that are declining or absent in natural water courses (Killeen, 1998; Painter, 1998; Armitage et al., 2003; Gómez and Araujo, 2008). Hudson and Harding (2004) identified 29 species of native fish that utilise New Zealand drains, several of which are threatened. Given the importance of drains to fish, both overseas and in New Zealand, anthropogenic disturbances in these systems most likely have a significant impact on regional biodiversity.

1.2 Macrophyte control

Proliferation of macrophytes in unshaded, eutrophic, low-gradient streams is detrimental to agriculture, and aquatic plant management is common in New Zealand (Garner et al., 1996; Kaenel, 1998; Hudson and Harding, 2004). In lowland areas the productivity of agricultural operations is dependent on the drainage capabilities of the surrounding waterways (Lalonde and Hughes-Games, 1997; Blann et al., 2009). Pasture growth is reduced in poorly drained areas (Schulte et al., 2006), and excess moisture must be removed from the soil quickly and efficiently to promote conditions that are favourable for agricultural production (Lalonde and Hughes-Games, 1997; Gibbs, 2006; Blann et al., 2009). Accelerated macrophyte growth associated with increased nutrient input can increase flow resistance within a waterway to the point that hydraulic capacity is reduced (Wilcock et al., 1999a; Luhar et al., 2008; Jones et al., 2012). Elevated water tables in

adjacent areas (Jones et al., 2008) then saturate the pasture increasing flood risk during periods of heavy rain or high flows (Hearne and Armitage, 1993; Kaenel, 1998). To maintain adequate drainage, macrophytes are regularly cleared from the streams that drain agricultural land, using herbicides, mechanical or manual extraction of plant material and, occasionally, plant-eating fish (Biggs and Close, 1989; Pieterse and Murphy, 1990; Wells et al., 2003; Hudson and Harding, 2004). Mechanical excavation and herbicide application are the most common forms of macrophyte control employed in New Zealand. Hudson and Harding (2004) calculated that local government agencies conduct these activities in more than 15,500 kilometres of water, which comprises approximately one thirtieth of New Zealand's total waterway length [425,000 kilometres (Van Bunnik et al., 2007)]. The total extent of mechanical excavation and herbicide application is expected to be much greater, as the majority of drain maintenance conducted by private land owners is not recorded (Hudson and Harding, 2004). It is vital that the ecological effects of widespread macrophyte removal on native fish are understood, as this activity may be a significant source of disturbance.

The perception that drains are of low ecological value, has limited the research focused on understanding and minimising the impact of macrophyte removal on fish. Garner et al. (1996) reported that decreased abundance of preferred invertebrate prey species following weed cutting in British streams significantly reduces the growth rates of young roach (*Rutilus rutilus*). Although Swales (1982) recorded significant short-term decreases in fish abundance following mechanical weed cutting in another British river, he was unable to isolate the effects of macrophyte removal from natural temporal fluctuations in population density. Consequently, the impacts of reduced food availability on fish population structure following macrophyte removal in British streams cannot be determined from the available literature. Population-level responses have been observed following macrophyte removal in the North-Atlantic region of the United States. Serafy et al. (1994) found that 10 % to 20 % of the resident fish population were removed from the channel during mechanical weed cutting in a Maryland river. The loss of such large numbers of fish significantly reduced abundance in the channel, but only in the short term. Fish were found to rapidly re-colonise reaches in which weed cutting was conducted, and 43 days after treatment fish density was higher in weed cut areas than in the control sites (Serafy et al., 1994). Although the population-level responses recorded in Serafy et al. (1994) were short-lived, it is likely that in New Zealand the effects of macrophyte removal are more persistent. The experimentally cleared areas in Serafy et al. (1994) were small (450 m^2) and bordered by undisturbed macrophyte beds. Recovery time after disturbance has been shown to decrease when undisturbed areas from which displaced animals can recolonise are close (Niemi et al., 1990; Reice et al., 1990). Therefore, in

New Zealand, where macrophyte removal is typically carried out on a much larger scale, fish populations would be expected to recover at a much slower rate than in Serafy et al. (1994) (Resh et al., 1988).

Despite the large scale on which macrophyte control is conducted in New Zealand, there has been little effort to quantify the ecological impacts of this activity. Anecdotal reports of the effects of macrophyte removal are confusing and counter-intuitive, suggesting that the abundance of environmentally sensitive species [inanga (*Galaxias maculatus)* and smelt (*Retropinna retropinna*)] is increased while the abundance of hardy species [common bully (*Gobiomorphus cotidianus)* and eels (*Anguilla* spp.)] is decreased (Goldsmith, 2000). In addition, the designs of previous experimental studies focused on quantifying modifications in community structure following macrophyte removal, have had insufficient power to detect changes in fish abundance or diversity (Goldsmith, 2000; Hudson and Harding, 2004; Young et al., 2004). Consequently, the impacts of macrophyte removal on New Zealand's native fish are unclear. A major objective of this thesis research was to determine the effects of macrophyte removal on native fish communities in order to establish the role of this activity in the disturbance of drain ecosystems.

Given the importance of macrophytes in aquatic environments (Fox, 1992), it is likely that their absence following macrophyte removal is a significant source of disturbance affecting fish communities. Macrophytes increase habitat complexity and provide cover and spawning habitat for many fish species (Mortensen, 1977; Garner et al., 1996). Furthermore, increased cover (Gregg and Rose, 1985) and food (Carpenter and Lodge, 1986; Kaenel, 1998; Cortelezzi et al., 2013) increases the availability of invertebrate prey in dense macrophyte stands (Voigts, 1976; Gregg and Rose, 1985; Carpenter and Lodge, 1986; McAbendroth et al., 2005). Consequently, the immediate loss of a high proportion of the available plant cover (Kaenel and Uehlinger, 1998) not only homogenises the stream bed, thereby limiting the number of fish species it can support (Hicks and Reeves, 1994), but reduces food availability (Garner et al., 1996). Macrophytes also influence oxygen cycles (Walling and Webb, 1992) and sediment deposition and transport (Luhar et al., 2008; Jones et al., 2012) in lotic environments. Therefore, changes in sediment regime and dissolved oxygen (DO) concentrations resulting from mechanical macrophyte removal and herbicide application may have a greater impact on resident fish than reduced plant cover.

1.3 Suspended sediment

Aquatic plants increase the deposition and retention of easily-suspended fine sediment (Luhar et al., 2008; Jones et al., 2012), yet it is unclear how mechanical excavation of macrophyte stands affects suspended sediment (SS) in drains. Brookes (1986) attributed sediment deposition downstream from drainage works in English waterways to increased sediment resuspension, but did not quantify changes in SS concentration in the water column. Wilcock et al. (1998) and Young et al. (2004) reported that sediment resuspension following excavation of New Zealand waterways significantly increases turbidity in the short term, but has no lasting effects. The rapid recovery of turbidity after excavation in these studies has led to suggestions that increased SS most likely has minimal impact on aquatic ecosystems after excavation (Wilcock et al., 1998; Hudson and Harding, 2004; Young et al., 2004). However, experimental excavation undertaken by both Wilcock et al. (1998) and Young et al. (2004) was limited to short sections of waterway. Consequently, the results of these studies may not reflect patterns of sediment resuspension following large-scale macrophyte control operations in which entire waterways or catchments are excavated.

Mechanical excavation of large sections of waterway most likely causes significant long-term increases in SS. Macrophytes have been shown to be important regulators of sediment resuspension (James and Barko, 1994; Madsen et al., 2001; James et al., 2004; Jones et al., 2012). Bed shear-stress increases with decreasing macrophyte density, and significantly more sediment is suspended by water movement in plant-free areas than in dense macrophyte stands (James and Barko, 1994; Madsen et al., 2001; James et al., 2004; Jones et al., 2012). Consequently, it is likely that reduced bed stability after large macrophyte removal operations will result in the continual resuspension of unconsolidated sediment by fluvial processes. SS concentrations after mechanical excavation of macrophytes are likely to remain elevated until all disturbed material is transported out of the system, or sediment compaction and new macrophyte growth increases the retention of deposited material.

Understanding how mechanical excavation affects sediment resuspension is vital, as elevated SS can reduce fish abundance and alter community structure in freshwater environments (Redding et al., 1987; Boubée et al., 1997; Brown et al., 1998; Lake and Hinch, 1999; Robertson et al., 2007; Crosa et al., 2010; Kemp et al., 2011). Boubée et al. (1997) determined that SS concentrations needed to reduce recruitment in New Zealand streams ranged from approximately 22 mg L^{-1} to >1885 mg L^{-1} [25 to 1,100 nephelometric turbidity units (NTU) in (Boubée et al., 1997) = approximately 21.3-1885.4 mg L^{-1}] amongst native fish species. In addition, there is evidence to

suggest that the regular occurrence of SS concentrations over 120 mg L^{-1} can reduce the abundance of sensitive species such as banded kokopu (*Galaxias fasciatus*) and redfinned bullies (*Gobiomorphus huttoni*) (Rowe et al., 2000). Therefore, if macrophyte removal leads to persistent increases in SS concentration, recruitment and abundance of these species is likely to decrease (Richardson and Jowett, 2002). In this thesis, the effects of large-scale macrophyte removal on suspended sediment dynamics were evaluated, in an effort to better understand the impact this activity has on fish health.

Sediment is an important source of pollution in aquatic ecosystems, and long-term increases in SS following mechanical excavation of macrophytes may be detrimental to the health of freshwater fish (Alabaster and Lloyd, 1982; Lazar et al., 2010; Kemp et al., 2011). Suspended sediment binds to and abrades delicate gill structures (Lake and Hinch, 1999; Sutherland and Meyer, 2007), and reduced respiratory performance is a commonly cited effect of increased SS on post-hatching life stages (Bruton, 1985; Henley et al., 2000; Bilotta and Brazier, 2008; Kemp et al., 2011). Physiological stress responses, such as reduced growth rates (Sutherland and Meyer, 2007), increased hematocrit production and decreased leukocrit production (Lake and Hinch, 1999), in high concentrations of suspended sediment have been linked with gill damage in spotfin chub (*Erimonax monachus*) (Sutherland and Meyer, 2007) and coho salmon (*Oncorhynchus kisutch*) (Lake and Hinch, 1999). If sediment inflicts the same level of gill damage in native species, reduced respiratory performance may affect fish health following mechanical excavation of macrophytes in New Zealand drains. Quantification of the effects of the sediment concentrations recorded during and after macrophyte removal on the respiratory performance of fishes in New Zealand drains, was an objective of this thesis.

The effects of suspended sediment at lower trophic levels may also have significant impacts on fish growth and community structure after macrophyte removal (Henley et al., 2000; Kemp et al., 2011). Physical abrasion and reduced light penetration at high SS concentrations can reduce periphyton and macrophyte abundance (Bruton, 1985; Van Nieuwenhuyse and LaPerriere, 1986; Graham, 1990; Davies-Colley et al., 1992), thereby limiting food availability to macroinvertbrates (Henley et al., 2000; Kemp et al., 2011). This, combined with increased drift as invertebrates are dislodged by sediment, can reduce abundance (Quinn et al., 1992; Wood and Armitage, 1999; Kemp et al., 2011). Reduced invertebrate abundance caused by increased sediment suspension can hinder the ability of some fishes to meet metabolic energy demands, as macroinvertebrates are an important dietary component of many species (Quinn et al., 1992; Wood and Armitage, 1999; Kemp et al., 2011). Furthermore, reduced visibility impairs the ability of visual foragers to locate

and obtain food resources in high concentrations of SS. Reactive distance to prey decreases with increasing SS as does capture success, which increases energy expenditure during feeding (Sigler et al., 1984; Sutherland and Meyer, 2007; Hazelton and Grossman, 2009a; Kemp et al., 2011). In New Zealand streams SS concentrations over 160 mg L^{-1} [160 in Quinn et al. (1992) = approximately 140-160 mg L^{-1}] reduce the availability of invertebrate prey (Quinn et al., 1992), but it is unclear how SS concentrations above this level affect the ability of adult New Zealand fishes to locate and obtain reduced food resources. The effects of the SS concentrations recorded during and after macrophyte excavation on the feeding performance of fish in New Zealand drains were examined in this thesis.

Even short-lived increases in SS following excavation may have population-level effects on fish in New Zealand drains. Rowe et al. (2009) determined that the SS concentrations required to cause 50 % mortality during 24 hours of exposure ranged from 3000 mg L^{-1} to greater than 43,000 mg L^{-1} amongst sensitive New Zealand fishes. SS concentrations as high as this are extremely rare in New Zealand streams (Hicks et al., 2004), and are unlikely even during excavation. However, changes in water chemistry associated with sediment resuspension after mechanical excavation may cause rapid mortality at much lower concentrations than those presented in Rowe et al. (2009).

Sediment suspended during macrophyte removal may also have significant impacts on fish communities when deposited in downstream receiving environments. Deposited fine sediment has been found to directly affect fish by smothering developing eggs (Kemp et al., 2011), reducing oxygen availability near the benthos (Bruton, 1985; Henley et al., 2000) and altering habitat suitability and availability by infilling interstitial spaces (Walling and Amos, 1999; Collins and Walling, 2007), reducing the availability of plant cover (Yamada and Nakamura, 2002) and changing riffle-pool sequence structure (Berkman and Rabeni, 1987). Fish communities may also be affected by the response of lower trophic levels to sediment deposition after macrophyte removal (Henley et al., 2000; Kemp et al., 2011). Deposited fine sediment can reduce food and benthic habitat availability to invertebrates by smothering periphyton and macrophytes (Brookes, 1986; Graham, 1990; Ryan, 1991; Yamada and Nakamura, 2002) and infilling interstitial spaces (Walton et al., 1977; Kemp et al., 2011; Burdon et al., 2013). In addition, fine sediment accumulation can affect invertebrates directly by reducing oxygen supply near the benthos (Crosthwaite et al., 2008; Sear et al., 2008) These effects alter invertebrate prey availability to fish (Wood and Armitage, 1999; Matthaei et al., 2006), which will likely affect growth rates and community structure (Henley et al., 2000; Kemp et al., 2011). Time and budgetary constraints meant that the impacts of macrophyte removal on sediment deposition and fish communities in

downstream receiving environments could not be quantified in this thesis. Suspended sediment was judged to pose a more immediate and quantifiable threat to fish after macrophyte removal, and consequently, became the focus of this study.

1.4 Dissolved oxygen

Changes in DO during and after chemical and mechanical macrophyte control may cause significant fish mortality in New Zealand drains. Oxygen uptake by fish is dependent on external oxygen conditions, and the supply to body tissues is reduced in hypoxic environments. Sensitivity of fish to DO concentrations is species dependent (Alabaster and Lloyd, 1982), and the DO concentrations required to cause 50 % mortality during 48 hours of exposure range from 0.54 to 2.65 mg L^{-1} amongst New Zealand fishes (Landman et al., 2005). Although there is some disagreement about the relative sensitivities of different species, the results presented by Landman et al. (2005) and Dean and Richardson (1999) suggest that extended exposure to DO concentrations below 30 % saturation and acute exposure to concentrations below 10 % saturation during macrophyte removal in New Zealand streams will result in significant fish mortality [values extrapolated from temperatures and DO concentrations (mg L^{-1}) presented by Dean and Richardson (1999) and Landman et al. (2005)]. Given the importance of DO to fish health, it is vital that the effects of mechanical excavation of macrophytes and herbicide application on oxygen conditions in New Zealand drains are understood.

1.4.1 Mechanical excavation

Sediment resuspension during mechanical excavation of macrophytes may deplete oxygen to such an extent that fish abundance is reduced. Accretion of fine sediment in macrophyte beds promotes anoxic conditions in the lower layers of the stream bed by limiting oxygen exchange from the water to the top two to five millimetres of substrate (Simpson et al., 1998). Microbial decomposition in anoxic sediment mineralises and reduces particulate organic matter to soluble intermediates (DiToro, 2001; Krevs and Kucinskiene, 2012). During sediment resuspension, oxidisation of these intermediates reduces DO in the water column (DiToro, 2001; Waterman et al., 2011; Krevs and Kucinskiene, 2012). When determining lethal SS concentrations for native fish, Rowe et al. (2009) controlled for the effect of oxygen depletion by aerating the sediment. This may have led to the underestimation of the effects of SS on fish mortality. Resuspension of anoxic sediment in concentrations well below the lethal limits presented in Rowe et al. (2009) [12,860 mg L^{-1} (Bruton, 1985)] has been found to completely deoxygenate Lake Chilwa in Malawi, resulting in large fish kills (Bruton, 1985). The effects of sediment suspension on DO are likely to

differ between lotic and lentic systems. However, it is possible that where anoxic bed sediments are present, oxygen consumption by organic material following mechanical excavation of macrophytes may cause fish mortality at lower SS concentrations than the lethal limits suggested by Rowe et al. (2009). An objective of this thesis was to develop greater understanding of the effect of sediment resuspension during mechanical excavation on oxygen conditions in New Zealand streams.

1.4.2 Herbicide application

Herbicide application may also deplete DO to the extent that fish mortality is increased. Photosynthetic oxygen supply is reduced during and after plant death (Brooker and Edwards, 1973; Newbold, 1975), and biological oxygen demand is increased due to aerobic microbial decomposition of the vegetation (Jewell, 1971; Godshalk and Wetzel, 1978). In dense macrophyte stands atmospheric reaeration cannot offset the increase in oxygen consumption during decomposition, and complete deoxygenation can occur within a week of herbicide application (Jewell, 1971). Past studies have quantified the impact of herbicide application on DO in reservoirs and small lakes (Jewell, 1971; Brooker and Edwards, 1973; Newbold, 1975), but it is unclear how chemical macrophyte control affects the availability of oxygen to fish in New Zealand drains. In this thesis, the effects of macrophyte dieback following herbicide application on oxygen dynamics in New Zealand streams were evaluated.

1.5 Thesis structure

This thesis is structured as a series of stand-alone scientific papers for journal publication. As a result, some repetition may be apparent in closely related chapters. Internal referencing has been included in order to present the research as a coherent thesis. In Chapter 2, the effects of complete and partial mechanical excavation of macrophytes on fish behaviour, abundance and species diversity in Southland drains were quantified using surveys and radiotelemetry. The role macrophyte removal plays in the disturbance of drain ecosystems was then assessed, and the potential to reduce disturbance through partial excavation (limited to alternating 50-metre sections of drain) examined. The rest of the thesis built on the results of Chapter 2, and was focused on identifying the physico-chemical drivers of altered community structure following macrophyte removal.

In the second data chapter (Chapter 3), the effects of macrophyte removal on SS were examined. The immediate effects of mechanical excavation of macrophytes on SS concentrations were

quantified from water samples taken during experimental macrophyte removal in a Southland drainage network. Water sampling and continual turbidity monitoring were also conducted during a large macrophyte removal operation in the same catchment to determine the long-term impact of mechanical excavation on sediment resuspension. The potential for SS to reduce fish abundance after macrophyte removal was assessed in Chapter 4. Feeding trials and respirometry were used to determine how SS concentrations recorded after excavation (in Chapter 3) affects the respiratory performance and feeding ability of an introduced salmonid [brown trout (*Salmo trutta*)] found in New Zealand drains.

The effects of sediment resuspension during macrophyte removal and plant dieback after herbicide application on DO were examined in Chapter 5. Oxygen loggers were used to continually monitor DO in a number of Waikato streams before and after herbicide application and mechanical excavation of macrophytes. Comparisons of the frequency and persistence of lethal DO concentrations before and after macrophyte control were used to determine the potential for these activities to reduce DO to the extent that fish abundance is decreased.

In the final chapter, primary findings from this research are integrated, and the potential for disturbance during macrophyte removal discussed. Possible drivers of disturbance are identified, and the results examined from a management perspective.

Chapter 2.0: Complete versus partial macrophyte removal: The impacts of two drain management strategies on freshwater fish in lowland New Zealand streams[1]

[1] A paper based on this chapter has been published in Ecology of Freshwater Fish (Volume 21, 510-520)

2.1 Abstract

Complete macrophyte removal to maintain drainage performance in lowland streams can have a negative effect on resident fish communities, but few studies have quantified this impact. Moreover, limited research has been carried out exploring alternative approaches for macrophyte removal that minimise the impact on the resident fish community. The aims of this study were: (1) to determine how the current practice of removing almost 100 % of available macrophyte cover affects native fish populations in lowland New Zealand streams, and (2) to see if this impact can be reduced by limiting macrophyte removal to alternating 50 meter sections of the waterway. Native fish populations were surveyed before and after experimental macrophyte removal for the following three treatments: 1 - complete macrophyte removal. 2 - macrophyte removal from alternating 50-m reaches, 3 - control with no macrophyte removal. Radiotelemetry was used to monitor the behavioural response of individual giant kokopu (*Galaxias argenteus*) to the different treatments. The results of this study suggest that current macrophyte removal practices reduce catch per unit effort of fish by 60 %. Although limiting mechanical excavation of macrophytes to alternating 50-meter sections still had significant community level impacts, large giant kokopu did benefit from this strategy. All tagged giant kokopu remained in streams reaches partially cleared of macrophytes while in completely excavated reaches all individuals were displaced. These results demonstrate the threat current drain management practices pose to New Zealand native fish, and highlight the value of trialling alternative methods of macrophyte removal.

2.2 Introduction

2.2.1 Anthropogenic disturbance

Anthropogenic disturbance of streams draining agricultural and industrial land has reduced both the abundance and the range of native fish species globally (Maitland, 1995). The negative effects of dams and chemical pollution are well documented (Alabaster and Lloyd, 1982; Maitland, 1995; Santos et al., 2006; Zhai et al., 2010), but relatively little is known about the effects of many other human disturbances on freshwater ecosystems. For effective conservation management of lotic ecosystems, it is essential that anthropogenic threats to native fish are understood, as they often play key roles in aquatic communities. (Maitland, 1995). Thus, changes in fish abundance can have a disproportionately large effect on community structure (Maitland, 1995).

2.2.2 Impacts of macrophytes on drainage of low altitude pasture

A little understood source of disturbance in low-altitude rivers draining pastoral land is the regular removal of aquatic macrophytes (Swales, 1982; Young et al., 2004). For successful drainage of agricultural run-off, streams must remove water from the pasture quickly and efficiently while promoting physical and chemical conditions in the soil that are favourable for agricultural production (Hudson and Harding, 2004). Accelerated macrophyte growth, associated with increased nutrient input can limit drainage outfall (Armitage et al., 1994; Kaenel et al., 1998). High densities of these plants can increase sediment deposition, reduce flows and potentially flood the surrounding pastoral land (Hearne and Armitage, 1993; Kaenel and Uehlinger, 1998). To prevent this, it is necessary to regularly clear macrophytes from the streams that drain agricultural land, using herbicides, mechanical or manual extraction of plant material and, occasionally, plant-eating fish (Biggs and Close, 1989; Pieterse and Murphy, 1990; Wells et al., 2003; Hudson and Harding, 2004).

2.2.3 Impacts of macrophyte control on freshwater fishes

Although macrophytes limit the drainage efficiency of pastoral land they play a key role in maintaining desirable physical and chemical conditions in freshwater systems (Fox, 1992). It is therefore likely, that the removal of macrophytes reduces the functioning of aquatic ecosystems. In addition, the physical process of removing macrophytes can be a source of disturbance in itself. Herbicides may adversely affect non-target organisms and alter the chemical properties of the water column via the decay of aquatic plant material (Murphy and Barrett, 1990). Mechanical excavation is the most disruptive method of macrophyte control, and can result in the immediate

loss of a high proportion of the available plant cover (Kaenel and Uehlinger, 1998). This can reduce the heterogeneity of the stream bed thereby limiting the number of species it can support (Hicks and Reeves, 1994). Despite a widespread understanding that macrophyte removal probably has a detrimental impact on community structure in streams (Swales, 1982), its impact on freshwater fish populations has been the focus of relatively few studies. The limited research, however, has suggested that mechanical removal of macrophyte cover reduces growth rates and increases the predation of juvenile fish (Mortensen, 1977; Garner et al., 1996), leads to the removal of fish during mechanical clearing (Dawson et al., 1991; Serafy et al., 1994) and alters the behavioural patterns of many freshwater fish species (Swales, 1982).

To date, research on the effects of drain management on New Zealand's native fish populations has produced conflicting results. Goldsmith (2000) reported that there is anecdotal evidence that macrophyte removal increases the abundance of some fish species like inanga (*Galaxias maculatus)* and smelt (*Retropinna retropinna*), but leads to decreases in the abundance of other species like common bully (*Gobiomorphus cotidianus)* and eels (*Anguilla* spp.). In her own study Goldsmith (2000) found that chemical and mechanical macrophyte removal methods had no effect on the native fish assemblage of Southland streams. The validity of these results have since been questioned because of the limited power of the study and discrepancies in the methodology (Hudson and Harding, 2004; Young et al., 2004). As a result of the limited research on this topic, it is unclear how the widespread removal of frequently abundant macrophyte species like oxygen weed (*Egeria* spp.), starwort (*Callitriche* spp.), swamp willowweed (*Polygonum* spp.) and pondweed (*Potamogeton* spp.) affects New Zealand's native fish (Hudson and Harding, 2004).

One species of fish that may be particularly susceptible to macrophyte removal is the drift feeding giant kokopu (*Galaxias argenteus*). Macrophyte removal not only reduces the availability of diurnal cover for this fish, but also results in the removal of a large quantity of invertebrate prey from the water column (Hudson and Harding, 2004). Therefore, reduced food availability may increase the size of giant kokopu home ranges, increase levels of aggression and alter diel activity patterns (David and Closs, 2003; Hansen and Closs, 2005; 2009). Understanding how macrophyte removal impacts this species is particularly crucial given its "declining" status (Allibone et al., 2010).

2.2.4 Aims

The present study aimed to examine the community level impacts of mechanical macrophyte removal on native fishes, and the possibility of minimising these effects with an alternative

macrophyte removal approach. Specifically, I aimed to quantify the impacts of complete macrophyte removal on native fish communities, and contrast these changes with those from areas where macrophyte removal was limited to alternating 50-meter sections. It is predicted that fish abundance will be reduced by macrophyte removal, and that community composition will change. I also predict that these changes will be less severe when macrophyte removal is limited to alternating sections, and that large giant kokopu will be less likely to be displaced from streams cleared in this manner.

2.3 Methods

2.3.1 Study area

Waituna Lagoon is a largely unmodified coastal lake located approximately 40 kilometres south-east of Invercargill in the South Island of New Zealand (Figure 2.1). The lagoon's catchment consists of about 20,000 hectares of farmland, native forest, and the internationally important Awarua wetlands. Drainage input for the catchment is provided by three major streams that flow directly into the lagoon. Of these streams the Waituna Creek drains the largest area (12,500 ha), followed by Carran Creek (5,700 ha) and Moffat Creek (1,700 ha). The majority of waterways in the region are extensively modified as a result of agricultural development, and regular mechanical excavation of macrophytes is needed to maintain adequate drainage of the surrounding low lying pastoral land (Riddell et al., 1988). Despite regular macrophyte removal the fish community in the area is relatively diverse, and the catchment has a large population of the declining giant kokopu (Riddell et al., 1988; Thompson and Ryder, 2002; Allibone et al., 2010).

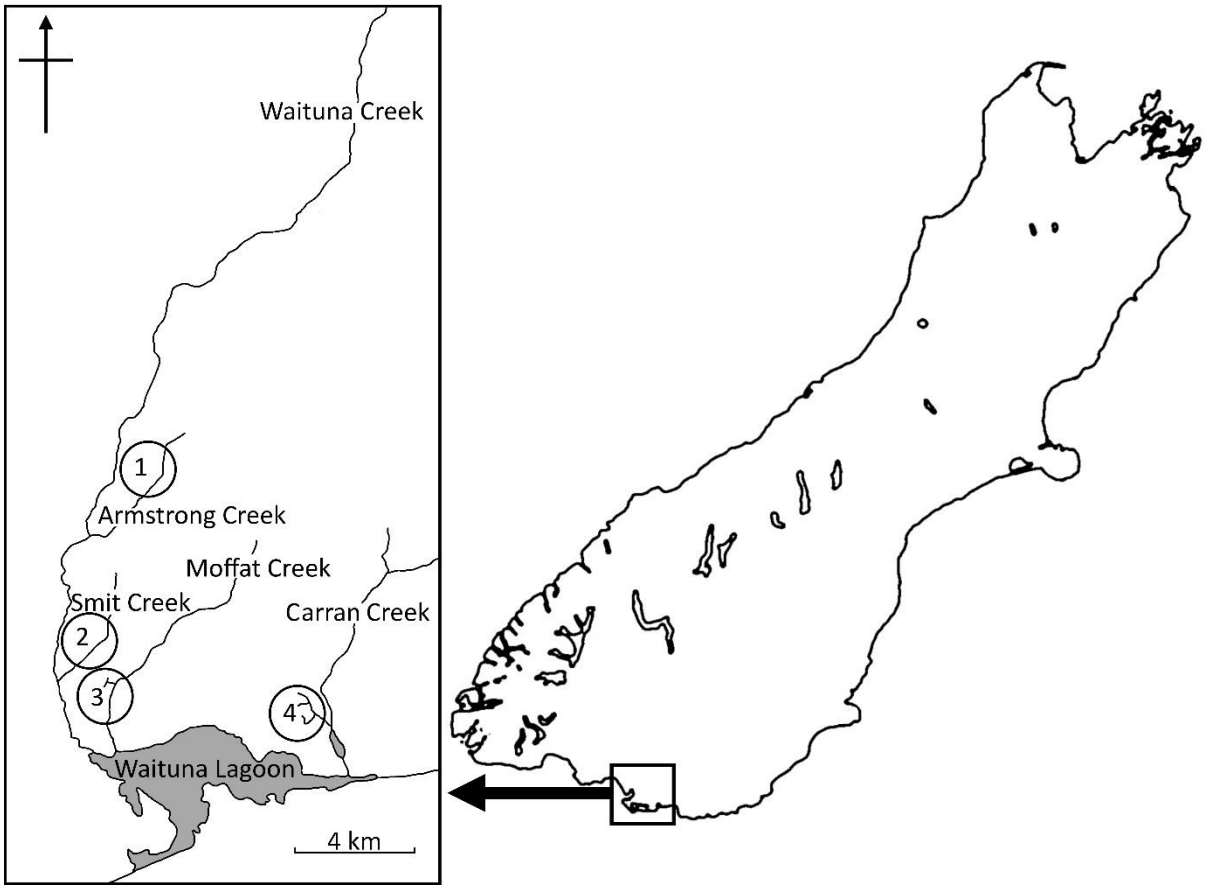

Figure 2.1 Study sites located in the reaches and tributaries of Armstrong Creek (1), Smit Creek (2), Moffat Creek (3), and Carran Creek (4), in the catchment of Waituna Lagoon in the South Island of New Zealand**.**

2.3.2 Study sites and treatments

The impacts of macrophyte removal were examined in 23 350-metre (m) treatment reaches located across four streams (Armstrong Creek, Carran Creek, Smit Creek and Moffat Creek). The decision to use 350-m study sites was based on the total length of waterway available to conduct the study and the number of replicates needed. Eight 350-m reaches were available along Moffat Creek, while five reaches were available in each of the remaining three streams. These 23 reaches were then randomly allocated to one of the following three treatments: 1 - complete macrophyte removal (cleared), 2 - macrophyte removal from alternating 50-m reaches - leaving half of the overall habitat intact (staggered), 3 - no macrophyte removal (control) (Figure 2.2).

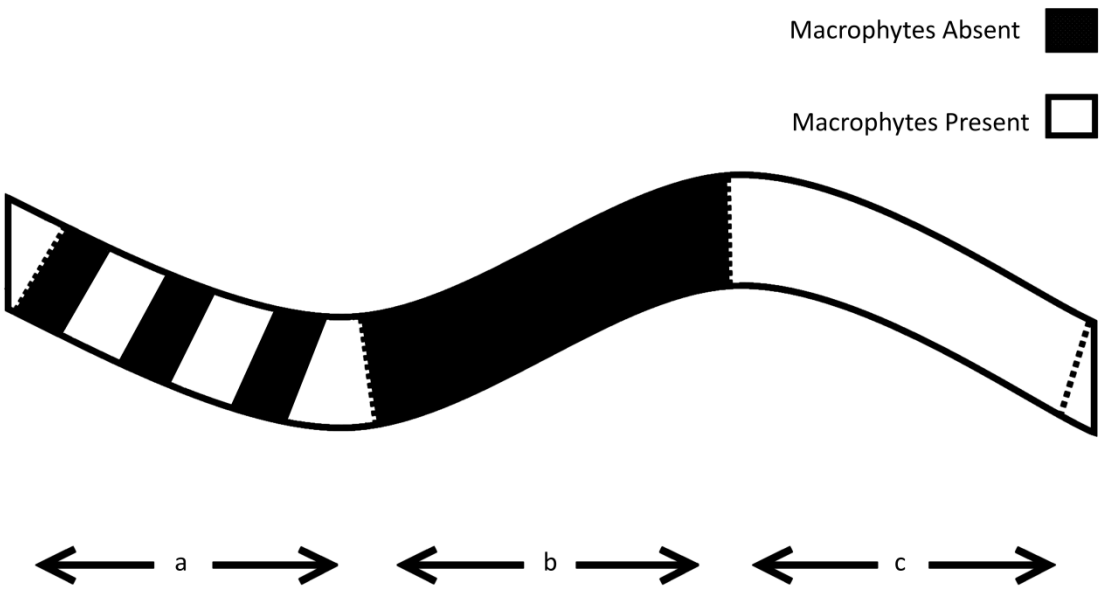

Figure 2.2 Illustration of the drain clearing treatments employed in this study. Section a = macrophyte removal limited to alternating 50-m sections (staggered); section b = complete removal of all aquatic macrophytes (cleared); section c = all macrophytes left undisturbed (control).

All reaches had > 50 % macrophyte coverage, and were all separated by at least 50 m of undisturbed waterway. This resulted in seven replicates of the cleared treatment (three in Moffat Creek, two in Armstrong Creek, one in Smit Creek, one in Carran Creek), eight replicates of the staggered treatment (three in Moffat Creek, two in Armstrong Creek, two in Smit Creek, one in Carran Creek) and eight control replicates (two in Moffat Creek, two in Armstrong Creek, one in Smit Creek, three in Carran Creek). Unfortunately, the different number of replicates of each treatment in each stream was unavoidable due to the wishes of various landowners and local government agencies. Experimental macrophyte removal was carried out between the 22/03/2011 and the 25/03/2011 using a mechanical excavator equipped with a perforated bucket that removed both plant material and silt from the streambed, while allowing water to flush back into the channel (Figure 2.3). Material removed from the waterway was placed on the bank immediately adjacent to the channel.

Figure 2.3 Photograph of excavator used in the study sites.

2.3.3 Physico-chemical properties and habitat structure

To determine the impact of the different macrophyte removal techniques on fish habitat structure, the physico-chemical characteristics of each treatment reach were analysed twice, once before excavation (between the 05/03/2011 and the 20/03/2011) and once after (between the 10/04/2011 and the 25/04/2011). Water temperature and relative conductivity were measured at the most upstream point of each reach using a YSI probe (YSI Inc. Model 85). Eight transects were then placed along the length of the reach at 50-m intervals. At each transect the percentage of the stream width covered by key plant groups (macrophytes, bryophytes, mat algae and filamentous algae) was estimated, and coverage was recorded on a scale of 1 to 3 (1 = rare i.e. < 20 % coverage; 2 = common i.e. 20-60 % coverage; 3 = abundant i.e. > 60 % coverage). Stream width was then measured at water surface level, and water depth was measured at 0.5-m intervals across the transect. Transect data were compiled, and used to estimate the maximum depth, mean depth and mean width of each reach as well as the mean abundance scores of key aquatic plant groups.

The percentage of the stream width covered by the following substrate types: mud [< 1 millimetre (mm)], sand (1-2 mm), fine gravel (3-20 mm), coarse gravel (21-60 mm), cobble (61-260 mm) was visually assessed at each transect. A total of 20 particles within each substrate category were collected from four points picked haphazardly along each transect, and measured across the intermediate axis, excluding mud and sand which were assumed to be 0.5 mm and 1.5 mm respectively. From these values the mean size of particles belonging to each substrate category was estimated, and used in the following formula to calculate overall mean particle size at each transect.

$$Mean\ particle\ size = \frac{\sum_{i=1}^{n}(S_t \times P_t)}{100}$$

Here *S* represents the mean particle size of substrate type *t*, and *P* represents the percentage of the width of the stream covered by *t*. Substrate size was averaged across the eight transects to calculate a mean value for each treatment reach.

2.3.4 Fish population surveys

The impacts of the different macrophyte removal methods on fish abundance and diversity were measured by surveying the fish populations of each study reach twice, once before excavation (between the 05/03/2011 and the 20/03/2011) and once after (between the 10/04/2011 and the 25/05/2011). Treatment reaches were separated into four individual 50-m sampling sites, and the resident fish populations in each site sampled separately by overnight netting. To avoid catching non-resident fish the most downstream 50 m and the most upstream 100 m in each 350-m treatment reach were excluded from sampling. Large fish (> 130 mm in length) were captured using two plastic fish traps per 50-m sampling site. The traps were constructed from cylinders of plastic netting (mesh size 20 mm), and measured 1.2 m x 0.40 m. Funnelled vertical entrances constructed from plastic netting allowed fish to enter the trap from either end, even in water too shallow for the use of conventional fyke nets (Figure 2.4). The traps were set approximately 20 m apart, positioned parallel to the bank, and secured to the stream bed using metal pegs. Smaller fish (< 130 mm) were caught using g-minnow traps that measured 420 mm x 230 mm, had 25 mm entrance apertures and a mesh size of 2 mm (Swales, 1987). Three g-minnows, secured to one another with a length of nylon rope, were set in the centre of each 50-m sampling site, and positioned so the long axis ran parallel to the bank.

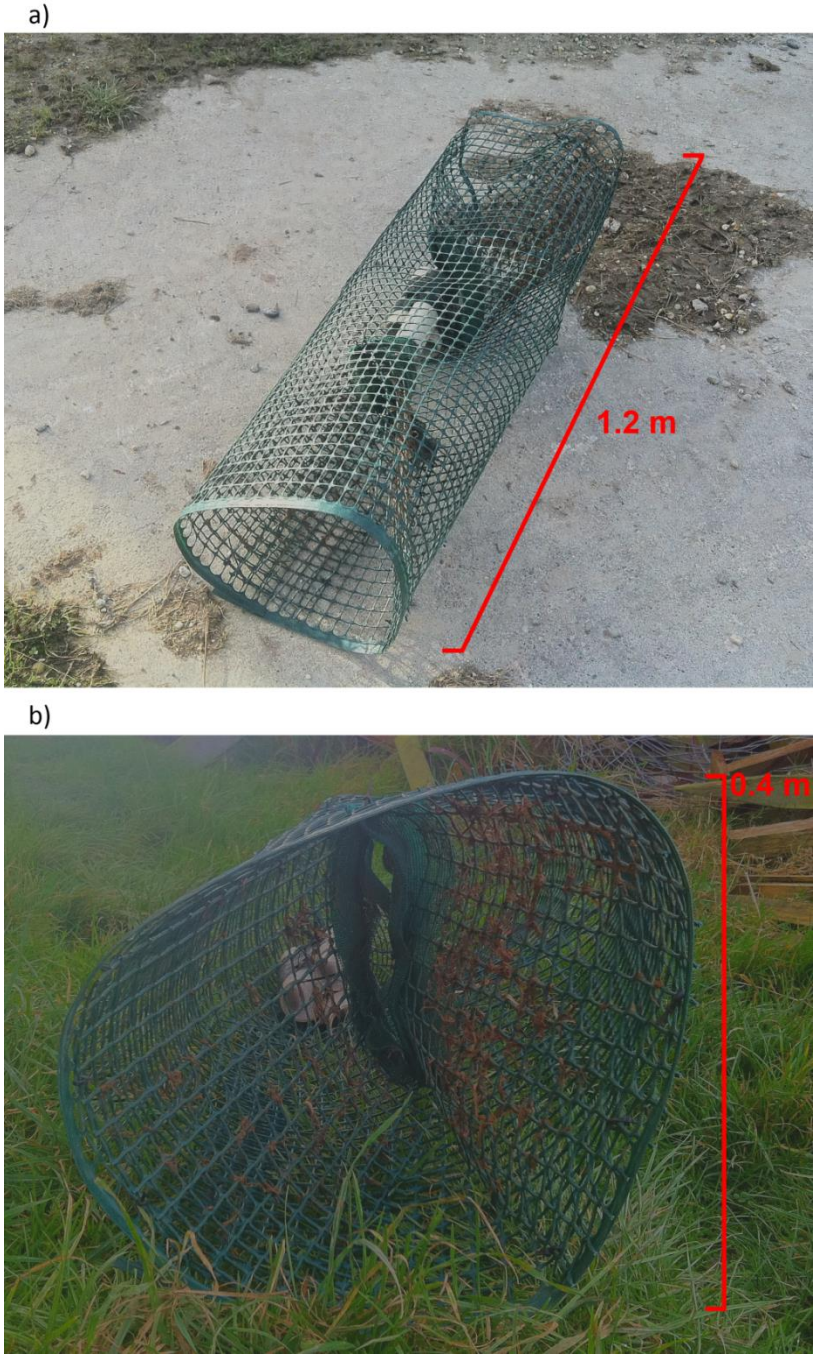

Figure 2.4 Photograph of the plastic fish traps used to catch Large fish (> 130 mm in length) during fish surveys. a) trap viewed from above; b) trap viewed from the end. Note PVC pipes were placed inside the traps to provide cover.

All fish traps were deployed between one and three hours before dusk, and collection began an hour after dawn. Captured fish were identified to the species level, and released at the point of capture immediately after examination (except large giant kokopu kept for radio tagging). The abundance of individual species in each 50-m sampling site was recorded in CPUE (catch per unit effort), i.e., the total number of fish caught in all nets per hour of fishing (fish/hour). As the catch

rates of individual species were low, the CPUE data from all species present at a site were pooled to produce a satisfactory estimate of total fish abundance (Swales, 1982). Fish collection was approved by the University of Otago Animal Ethics Committee, and conducted in accordance with the University of Otago Code of Ethical Conduct.

2.3.5 Radiotelemetry

To measure the response of individual giant kokopu to the different macrophyte removal techniques, radiotransmitters (ATS model F1030) were surgically implanted into the abdominal cavities of 11 large adult giant kokopu (three from Armstrong Creek, four from Moffat Creek and four from Carran Creek). Four fish were captured in the first sampling period between the 08/03/2011 and the 21/03/2011, and the remaining seven fish were collected between the 22/03/2011 and the 25/03/2011 after being removed from the waterways by the excavator during macrophyte removal. Eight fish were tagged in reaches from the cleared treatment group, two from the staggered and one from the control. The limited numbers of fish caught from within the stream (i.e. not with the excavator) resulted in low numbers of fish being tagged in staggered and control reaches (Table 2.1). Further surveys were carried out to increase the numbers of fish in the control reaches, but no fish were caught.

Radio implants were carried out following the methodology of David and Closs (2001). Briefly, fish were anaesthetised with a dilute solution of AQUI-S (20 µl l^{-1}), and placed ventral side up in a fish shape relief cut into a wet sheet of foam rubber. A 10 mm incision was made approximately 15 mm posterior to the base of the pelvic fins and between three to five mm to the right of the ventral midline. A radiotransmitter was inserted into the fish's abdominal cavity through this incision. The fish was kept unconscious during surgery by aerating the gills with a dilute solution of AQUI-S (20 µl l^{-1}). The incision was flushed with saline solution and closed with three interrupted stitches. To reduce the risk of infection and decrease healing time the wound was also coated with a povidone-iodine topical antiseptic (Betadine, Purdue Pharma).

The transmitters weighed 2.1 grams (in air), had an internal loop transmitter and an estimated battery life of 94 days (30 pulses min^{-1}, pulse width 15 milliseconds). Based on Winter (1983) recommendation the transmitters were no more than 2 % of the body weight of the fish into which they were implanted. The mean weights and standard lengths (SL = snout to base of caudal fin) of tagged fish were 186.3 mm and 142.7 g (Armstrong Creek), 214.9 mm and 195.9 g (Moffat Creek), and 214.4 mm and 195.1 g (Carran Creek) (Table 2.1).

Following surgery the fish were left to recover for between 10 and 20 minutes, and then released at the point of capture once equilibrium had been regained. Fish were located twice in the 48 hours following surgery to ensure the tags were not expelled. As in David and Closs (2003) location data were not recorded for a minimum of two weeks after tagging to ensure behavioural patterns had returned to normal. Radio-transmitters were not recovered from tagged fish at the end of the study. Surgical methods were approved by the University of Otago Animal Ethics Committee, and conducted in accordance with the University of Otago Code of Ethical Conduct.

Telemetry data collection commenced on the 13/05/2011 and ended on the 11/06/2011. When weather conditions permitted, each fish was tracked twice every 48 hours, once during the day (09:00 -17:30 hours) and once at night (18:30 -03:00 hours). Fish were located using a scanning receiver (Falcon five, Wildlife Materials International Inc., Carbondale, IL, USA) and a hand-held directional three element Yagi antennae. The positions of located fish were recorded as the distance and direction (either upstream or downstream) of their current location in relation to where they were first located and released following tagging. If a fish was located in a study reach the treatment group of the reach was recorded. If the reach belonged to the staggered treatment group, it was noted whether the fish was in the cleared or undisturbed areas. Retrieval of concealed transmitters prior to data collection indicated that estimated positions were accurate to ±0.5 m. Fish that could not be located within 1500 m of where they had been tagged over four days of attempted tracking were considered to have left the study area. Additional checks to locate them were then undertaken once each week after this four day period.

Table 2.1 Weights and standard lengths of individual fish, whether it was located in the study area after macrophyte removal, the total number of day and night locations and the dates between which location data was collected

Stream	Fish	Located	L (mm)	W (g)	Treatment	Day	Night	Track Period
Armstrong	A176	No	190.5	118.9	Cleared	-	-	-
	A296	Yes	180.9	142.6	Cleared	1	-	26/5-27/5
	A127	No	187.5	166.5	Cleared	-	-	-
	Mean	-	186.3	142.7	-	-	-	-
Moffat	M176	Yes	231.0	244.8	Control	2	2	13/5-26/5
	M299	Yes	198.1	135.9	Staggered	5	7	13/5-12/6
	M127	Yes	195.6	162.8	Cleared	2	2	13/5-26/5
	M266	Yes	235.0	240.0	Staggered	5	6	13/5-26/5
	Mean	-	214.9	195.9	-	-	-	-
Carran	C156	Yes	221.4	234.8	Cleared	2	-	14/5-16/6
	C286	No	209.1	158.5	Cleared	-	-	-
	C226	Yes	198.9	156.7	Cleared	7	6	14/5-11/6
	C136	Yes	228.1	230.3	Cleared	1	-	14/5-15/5
	Mean	-	214.4	195.1	-	-	-	-

2.3.6 Analyses

2.3.6.1 Data exploration and transformation

Shapiro-Wilk tests were used test data for the assumption of normality before statistical analysis was conducted. Catch data were not normally distributed, and both CPUE and species data were log transformed ($\log_{x+1}$) to approximate normality. All statistical analyses were carried out using SPSS Statistical Software version 20.0.0 (International Business Machines Corporation, Armonk, NY, USA).

2.3.6.2 Fish catch data

Fish catch data collected from the same 50-m sampling site were not independent through time (Zar, 1984). Consequently, paired *t*-tests were used to compare total CPUE and changes in the

total number of species collected before and after the experimental macrophyte removal for each of the different treatment groups. The same tests were used to analyse changes in the combined and individual CPUE of common bullies and giant kokopu. Paired t-tests were used to compare CPUE before and after experimental macrophyte removal in uncleared and cleared sampling sites in the staggered treatment. To achieve this it was necessary to treat data from different 50-m sampling sites in the same treatment site as independent. Paired *t*-tests were used to compare the physical characteristics of each 350-m reach before and after macrophyte removal in each treatment group.

2.3.6.3 Radiotelemetry data

Statistical comparisons of the movement patterns of radio-tagged individuals from the different treatment groups were not possible due to small sample sizes and low levels of individual replication. Instead data are presented descriptively as in David and Closs (2003). The locations of radio-tagged fish after excavation in relation to where they were originally tagged were used to make inferences about the impacts of the different macrophyte removal techniques. The recorded positions of individual fish in relation to available macrophyte cover were plotted on a drawn-to-scale map of the study sites (Figure 2.7), and used for inferences about habitat use in staggered sites (David and Closs, 2003), and the potential benefits of this technique for giant kokopu management.

2.4 Results

2.4.1 Physico-chemical properties and habitat structure

After experimental macrophyte removal mean water temperature decreased from 13.58° Celsius to 11.25°C in cleared sites, from 14.06°C to 11.35°C in staggered sites and from 11.72°C to 10.98°C in control sites (Table 2.2). The observed difference in temperature before and after macrophyte removal was statistically significant in the cleared and staggered treatment groups but not in the control (paired *t*-tests, cleared (t_6 = 7.513, $P < 0.001$; staggered t_7 = 4.524, P = 0.003; control t_7 = 0.428, P = 0.682). In all three treatments relative conductivity, mean width, mean depth and mean substrate size did not differ significantly before and after macrophyte removal. Mean maximum depth was significantly shallower for cleared treatments, decreasing from 32.35 to 27.54 centimetres (paired *t*-test, t_6 = 8.304, P = 0.015), but depth did not change in staggered or control treatments (paired *t*-tests, staggered t_7 = 1.38, P = 0.21; control t_7 = -0.78, P = 0.46) (Table 2.2).

Mean macrophyte coverage was significantly reduced in both the treatment groups and the control group after macrophyte removal (paired *t*-tests, cleared t_6 = 2.00, P = 0.005; staggered t_7 = 4.492, P = 0.003; control t_7 = 3.416, P = 0. 01). The mean coverage score dropped from 2.43 to 1.14 at cleared sites, from 2.26 to 1.75 at staggered sites and from 2.71 to 2.25 at control sites (Table 2.2). There were no significant differences in the abundance of bryophytes, filamentous algae or mat algae before and after experimental macrophyte removal for any of the treatments.

Table 2.2 Physico-chemical parameters measured in cleared, staggered and control sites before and after experimental macrophyte removal.

	Cleared			
Parameters	Before Mean ± SE	After Mean ± SE	*t(df = 6)*	*P*
Temperature(°C)	13.58 ± 0.46	11.25 ± 0.58	7.51	<0.001*
Relative conductivity (µS/cm)	193.29 ± 14.70	211.74 ± 11.89	-1.19	0.278
Mean width (m)	1.36 ± 0.18	3.26 ± 1.51	1.84	0.261
Mean depth (cm)	24.25 ± 4.98	20.08 ± 3.37	8.80	0.07
Maximum depth (cm)	32.35 ± 4.36	27.54 ± 3.91	8.30	0.015*
Mean particle size (mm)	19.87 ± 10.92	11.9 ± 4.81	32.08	0.449
Macrophyte abundance	2.43 ± 0.21	1.14 ± 0.14	2.00	0.005*
Bryophyte abundance	1.32 ± 0.25	1.14 ± 0.14	0.52	0.253
Mat algae abundance	1.32 ± 0.25	1.05 ± 0.03	0.81	0.231
Filamentous algae	1.32 ± 0.21	1.07 ± 0.05	0.74	0.267
	Staggered			
Parameters	Before Mean ± SE	After Mean ± SE	*t(df = 7)*	*P*
Temperature(°C)	14.06 ± 0.56	11.35 ± 0.51	4.52	0.003*
Relative conductivity (µS/cm)	198.67 ± 12.54	206.73 ± 10.61	-0.46	0.657
Mean width (m)	1.41 ± 0.10	1.60 ± 0.15	-2.09	0.075
Mean depth (cm)	26.51 ± 3.83	26.01 ± 2.05	0.19	0.850
Maximum depth (cm)	36.92 ± 4.07	33.81 ± 2.32	1.38	0.210
Mean particle size (mm)	17.64 ± 7.07	15.91 ± 5.41	0.52	0.618
Macrophyte abundance	2.26 ± 0.16	1.75 ± 0.09	4.49	0.003*
Bryophyte abundance	1.15 ± 0.16	1.06 ± 0.06	1.00	0.351
Mat algae abundance	1.03 ± 0.03	1.03 ± 0.03	-	-
Filamentous algae	1.34 ± 0.23	1.00 ± 0.00	1.51	0.173
	Control			
Parameters	Before Mean ± SE	After Mean ± SE	*t(df = 7)*	*P*
Temperature(°C)	11.72 ± 1.56	10.98 ± 0.56	0.42	0.682
Relative conductivity (µS/cm)	208.04 ± 17.12	212.00 ± 12.03	-0.15	0.881
Mean width (m)	1.67 ± 0.21	2.45 ± 0.75	-0.99	0.351
Mean depth (cm)	29.40 ± 3.97	33.95 ± 4.41	-0.83	0.429
Maximum depth (cm)	38.82 ± 3.51	43.50 ± 5.39	-0.78	0.461
Mean particle size (mm)	26.75 ± 13.54	20.42 ± 7.81	0.50	0.605
Macrophyte abundance	2.71 ± 0.17	2.25 ± 0.25	3.41	0.011*
Bryophyte abundance	1.18 ± 0.10	1.12 ± 0.08	1.52	0.170
Mat algae abundance	1.09 ± 0.07	1.25 ± 0.14	-0.95	0.370
Filamentous algae	1.06 ± 0.06	1.00 ± 0.00	1.00	0.351

* Represents a significant difference ($P < 0.05$) in values observed before and after treatment

2.4.2 Fish population surveys

In total, 1250 native fish were collected from the 23 sites, 682 prior to experimental macrophyte removal and 568 after. Common bully made up the majority of the total fish catch (n = 1023). Giant kokopu (n = 140 and longfin eel (*Anguilla dieffenbachii*) (n = 69) were also relatively common, while inanga (n = 14), banded kokopu (*Galaxias fasciatus*) (n = 2), redfin bully (*Gobiomorphus huttoni*) (n = 1) and shortfin eel (*Anguilla australis*) (n = 1) were only found occasionally. The number of species caught from each replicate ranged from zero to four and was not found to differ significantly between the before and after macrophyte removal sampling periods for any treatments (paired t-tests, cleared $t_{27} = 1.148$, $P = 0.261$; staggered $t_{31} = 0.504$, $P = 0.618$; control $t_{31} = 1.082$, $P = 0.288$).

After macrophyte removal, mean CPUE significantly decreased in cleared treatments from 0.45 to 0.22 fish/hour (paired t-test, $t_{27} = 2.159$, p = 0.040). Similarly, CPUE was significantly reduced in staggered treatments from 0.52 to 0.32 fish/hour (paired t-test, $t_{31} = 2.088$, $P = 0.045$) (Figure 2.5a). CPUE did not differ before and after macrophyte removal for the control treatments (paired t-test, $t_{31} = -0.747$, $P = 0.461$) (Figure 2.5b). In staggered sites mean total CPUE significantly decreased in cleared areas from 0.51 to 0.206 (paired t-test, $t_{15} = 2.060$, $P = 0.049$) (Figure 2.6a). Mean CPUE in uncleared areas did not differ before and after macrophyte removal (paired t-test, $t_{15} = 0.760$, $P = 0.461$) (Figure 2.6b).

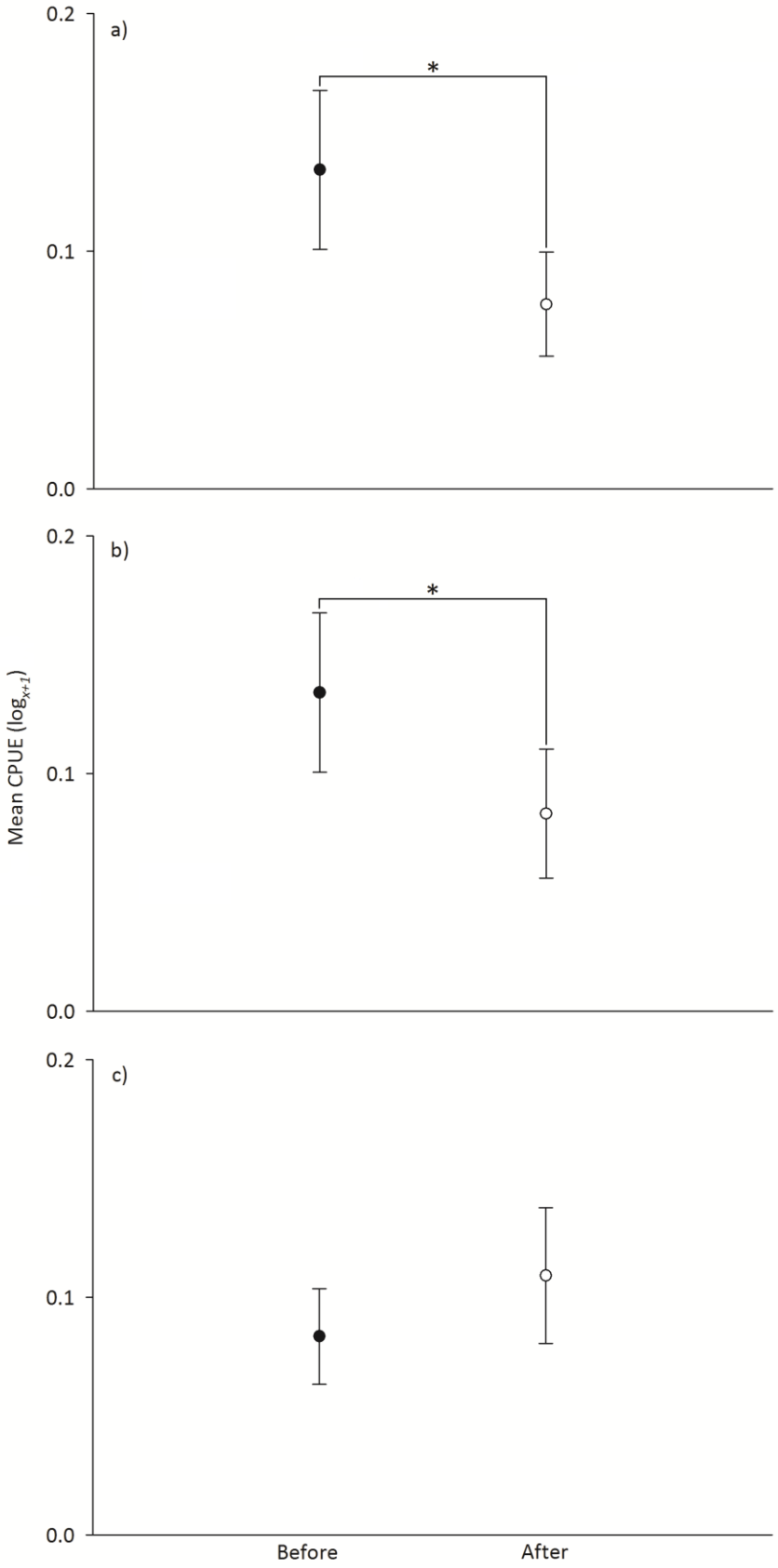

Figure 2.5 Mean (±SE) CPUE [fish caught per hour per 50 m (log_{x+1})] before (black circles) and after (white circles) macrophyte removal in the three treatment groups (a = cleared; b = staggered; c = control). Statistically significant differences in CPUE before and after macrophyte removal are illustrated with an *($P < 0.05$).

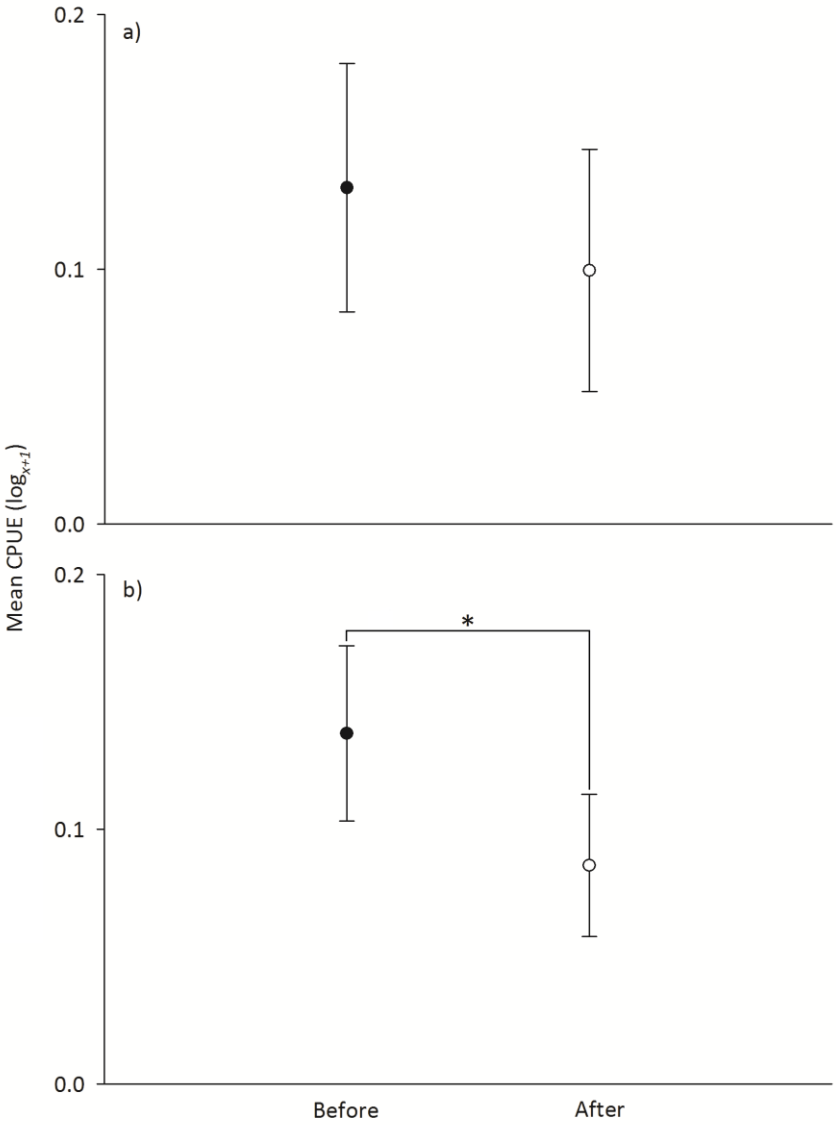

Figure 2.6 Mean (±SE) CPUE [fish caught per hour per 50 m (log_{x+1})] before (black circles) and after (white circles) macrophyte removal in the staggered treatment group (a = uncleared sections; b = cleared sections). Statistically significant differences in CPUE before and after macrophyte removal are illustrated with an *($P < 0.05$).

The combined CPUE of the two most abundant species, common bully and giant kokopu, significantly decreased from 0.48 to 0.21 fish/hour (paired t-tests t_{27} = 2.286, P = 0.030) in the cleared treatment after macrophyte removal. In contrast, combined CPUE did not differ significantly in either the staggered treatment or the control (paired t-tests, staggered t_{31} = 1.915, P = 0.065; control t_{31} = -1.383, P = 0.176 (Table 2.3). After macrophyte removal the CPUE of common bully only differed significantly in the cleared treatment, decreasing from 0.43 to 0.18 fish/hour (paired t-tests, cleared t_{27} = 2.104, P = 0.045; staggered t_{31} = 1.907, P = 0.066; control t_{31} = -1.764, P = 0.88) (Table 2.3). CPUE of giant kokopu did not differ significantly before and after macrophyte removal in either the treatments or the control (paired t-tests, cleared t_{27} = 1.311, P = 0.201; staggered t_{31} = 0.086, P = 0.932; control t_{31} = 1.014, P = 0.319) (Table 2.3).

Table 2.3 CPUE of common bully, giant kokopu and both species combined in cleared, staggered and control sites before and after experimental macrophyte

	Cleared			
	CPUE Before	CPUE After		
Species	Mean ± SE	Mean ± SE	*t(df = 27)*	*P*
Common bully	0.4281± 0.132	0.181 ± 0.062	2.104	0.045*
Giant kokopu	0.050 ±0.014	0.029 ± 0.012	1.311	0.201
Combined	0.478 ±0.134	0.2104 ± 0.067	2.286	0.030*
	Staggered			
	CPUE Before	CPUE After		
Species	Mean ± SE	Mean ± SE	*t(df = 31)*	*P*
Common bully	0.463 ±0.149	0.279 ±0.147	1.907	0.066
Giant kokopu	0.031 ±0.010	0.023 ±0.009	0.086	0.932
Combined	0.494 ±0.149	0.308 ±0.148	1.915	0.065
	Control			
	CPUE Before	CPUE After		
Species	Mean ± SE	Mean ± SE	*t(df = 31)*	*P*
Common bully	0.150 ±0.063	0.305 ±0.115	-1.764	0.088
Giant kokopu	0.071 ±0.018	0.054 ±0.013	1.014	0.319
Combined	0.222 ±0.063	0.360 ±0.113	-1.383	0.176

* Represents a significant difference ($P < 0.05$) in values observed before and after treatment

2.4.3 Radiotelemetry

Of the 11 fish originally tagged only three could not be located within the time frame of the study, all of which were from cleared treatments. Of the eight fish located at least once after excavation five were from the cleared treatments, two were from staggered treatments and one was from a control treatment. Fish were tracked between the 10/02/2011 and the 12/06/2011, with earliest contact lost on the 14/05/2011 (Table 2.1). Four of the five fish that were originally tagged in the cleared treatments left after macrophyte removal. M127 and C156 moved into control sites, C136 moved into a staggered site, and A296 moved upstream into an undisturbed area. In contrast, fish C226 was found to regularly use a completely cleared section of waterway at night (100 % of

night-time locations), but consistently moved upstream into a control site before dawn (100 % of day-time locations) (Figure 2.7a). This pattern was also seen in the movements of the two fish tagged in staggered sites. Both M299 and M266 used cleared sections of staggered sites at night (100 % of night-time locations), and returned to uncleared sections during the day (100 % of day-time locations) (Figure 2.7b & c). The single fish tagged in a control site, M176, remained in that site following macrophyte removal at the treatment sites.

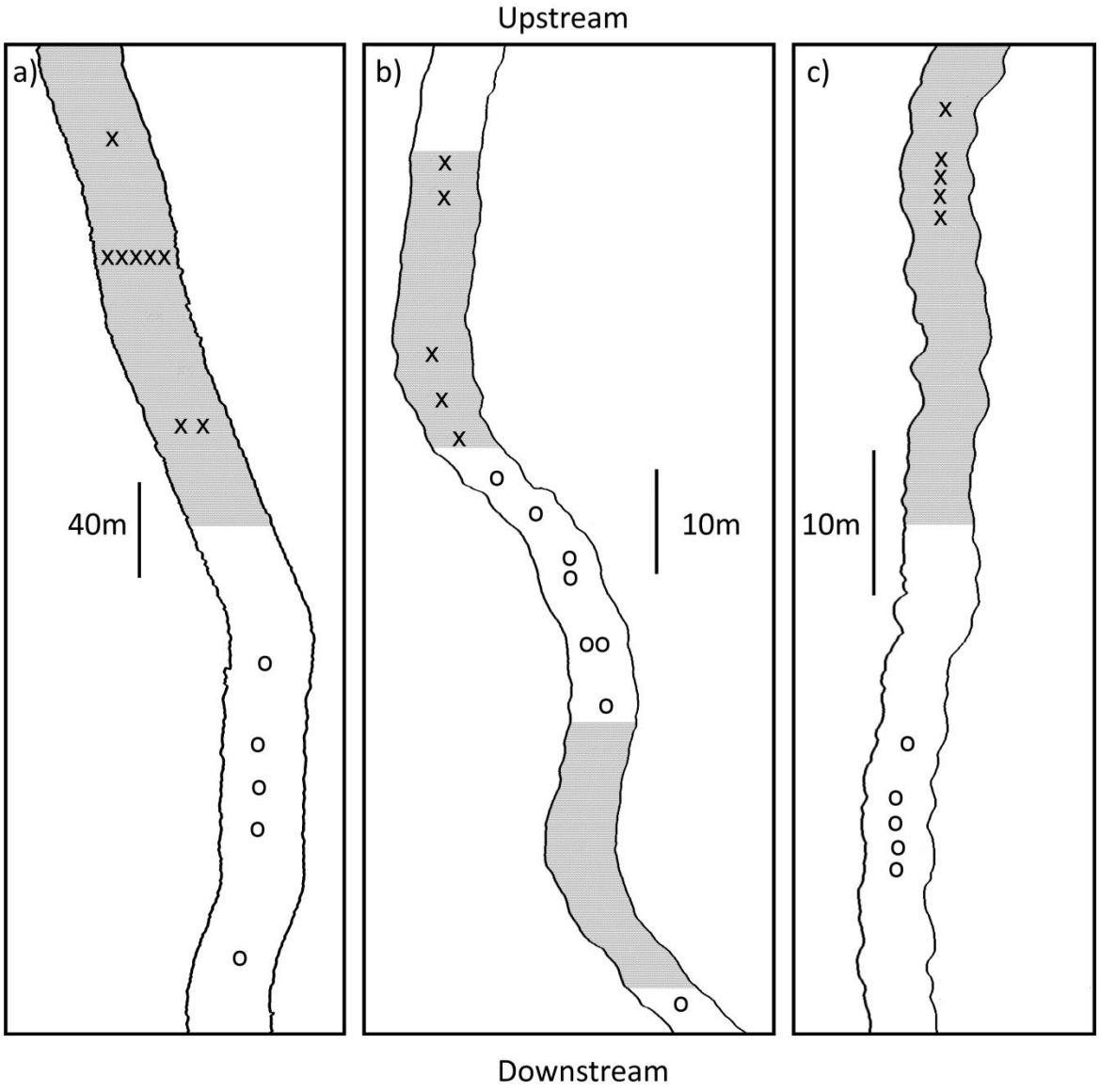

Figure 2.7 Point in time locations for three individual fish (a=fish C226; b= fish M299 c= fish M266). X= day cover positions and O = night positions. Shaded areas represent sections of waterway from which macrophytes were not removed. For ease of interpretation, channel widths are not drawn to scale.

2.5 Discussion

2.5.1 Impacts of macrophyte removal on native fish communities

2.5.1.1 Abundance and diversity

The results of this study suggest that complete macrophyte removal can significantly reduce total CPUE of native fish in lowland New Zealand streams. The reduction in combined CPUE of common bully and giant kokopu after macrophyte removal was statistically more significant than the decreases observed in either species individually. This indicates that macrophyte removal has the potential to reduce the abundance of both species, and that the lack of a significant change in giant kokopu abundance after excavation may be due to low statistical power associated with large variation within a small sample. Although there was no evidence to suggest that staggered macrophyte removal minimised the impacts on native fish abundance, results suggest that uncleared areas act as refuges for large giant kokopu which may be less likely to leave streams cleared in this manner. These findings support predictions that the complete clearing of streams negatively impacts freshwater fish populations, and that these impacts can be minimised by limiting macrophyte removal to alternating sections of a waterway.

The limited change in species richness observed after excavation may be a reflection on the long-term removal history of the study area rather than the effects of short-term macrophyte removal. Regular excavation of waterways has been carried out in the Waituna catchment since at least the 1960's (Johnson and Partridge, 1998), which may have generated localised fish communities that are resistant to macrophyte removal in areas that are frequently cleared. If so, fish assemblages in developed waterways, such as those where the current study was carried out, may only consist of species that are at least partially resistant to this form of disturbance (Orrego et al., 2009). Alternatively, change in species diversity may not have been detected because of sampling bias in the study design. Static traps are biased towards active and cover-seeking species (Portt, 2006). Consequently, any changes in community composition driven by species that are not easily trapped may not have been detected in the current study. Normally this could have been avoided through the use of active fishing methods such as spotlighting and electro-fishing. However, high turbidity and conductivity precluded the use of these techniques in this study. Determining how the distribution of native fish species relates to current and historical macrophyte removal activities in catchments like the Waituna may provide a better understanding of the community level impacts this activity has.

The design of this study did not allow measurement of the long-term impacts of macrophyte removal on the fish community. The experimentally cleared reaches included in this study were bordered by undisturbed macrophyte beds. Recovery time after disturbance has been shown to decrease when undisturbed areas from which displaced animals can recolonise are close (Niemi et al., 1990; Reice et al., 1990). Therefore, the fish populations in our study sites would be expected to recover much more rapidly than they would under real-world conditions where macrophyte removal is typically carried out over a much larger area (Resh et al., 1988). Future research focused on monitoring resident fish populations following routine large-spatial-scale macrophyte removal operations is needed to measure long term community responses to this activity accurately.

2.5.1.2 Benefits of adopting staggered approach

Although staggered clearing still resulted in significant reductions in native fish abundance this technique may reduce the effects of macrophyte removal on adult giant kokopu. Large giant kokopu are generally nocturnal, and seek cover when not active (Whitehead et al., 2002; David and Closs, 2003). Descriptive examination of radiotelemetry data collected in this study suggests that the response of giant kokopu to macrophyte removal is heavily dependent on the availability of macrophytes, and large individuals will leave completely cleared areas when diurnal concealment is no longer possible. In contrast, staggered macrophyte removal may preserve enough cover, and eliminate the need for large giant kokopu to leave treated waterways. Furthermore, this technique may actually benefit giant kokopu by increasing the availability of desirable nocturnal feeding habitat (David and Closs, 2003). Three individuals tracked in this study regularly moved from the densely vegetated areas in which they sheltered during the day to cleared areas at night. Giant kokopu prefer to feed in open habitats (David and Closs, 2003; Hansen and Closs, 2009), and by creating this habitat through staggered clearing, the numbers and condition of resident fish may remain unchanged despite reductions in food availability due to macrophyte removal (Garner et al., 1996). Although the idea of leaving undisturbed refuges when clearing macrophytes from streams has been suggested in the past (Swales, 1982; Armitage et al., 1994; Garner et al., 1996; Kaenel et al., 1998), this is the first attempt to quantify the actual benefits of this technique for individual fish species.

2.5.1.3 Importance of findings

The findings of this study are of particular importance as they demonstrate the threat posed by current drain management practices to native fish communities throughout New Zealand's lowland waterways, and suggest that the implementation of the staggered clearing strategy in areas where

the giant kokopu are found may prevent further losses of this already-threatened fish. Mechanical removal of macrophytes has been shown to lead to short-term decreases in fish abundance in both North American and British streams (Swales, 1982; Serafy et al., 1994), with limited impact on the species diversity of the resident fish communities (Serafy et al., 1994). It is commonly accepted that mechanical excavation of macrophytes in New Zealand streams most likely has a negative influence on native fish (Hudson and Harding, 2004), but the present study is the first to quantify this knowledge gap (Young et al., 2004)

2.5.2 Drivers of reduced fish abundance following macrophyte removal

2.5.2.1 Habitat loss

A common problem encountered with past macrophyte removal studies is the difficulty of isolating seasonal changes in abundance from treatment effects (Swales, 1982; Armitage et al., 1994). In this case, however, CPUE did not differ between sampling periods in the undisturbed control reaches despite seasonal decreases in temperature and significant reductions in macrophyte cover. This suggests that decreased fish abundance in cleared and staggered reaches was caused by the physical process of macrophyte removal, the resulting changes in habitat structure or a combination of both these factors. The total loss of available plant cover is, most likely, a contributing factor in the reductions in fish abundance seen in the two treatment groups. Removing large quantities of aquatic vegetation increases predation of smaller fish (Mortensen, 1977), reduces cover for adult fish (Swales, 1982) and decreases food availability for both predatory and herbivorous fish (Swales, 1982; Garner et al., 1996). Although staggered clearing still led to a significant decrease in total fish abundance, maintaining half of the available plant cover did mitigate at least some of the impacts of macrophyte removal. Undisturbed macrophyte beds acted as "refuges" for giant kokopu, and decreases in fish abundance were limited to the areas from which all available plant cover had been removed. Significant reductions in macrophyte coverage, however, had no impact on CPUE in the control reaches. Subsequently it is likely that changes in fish abundance in cleared and staggered reaches were influenced by a number of factors associated with macrophyte removal other than reduced plant biomass.

2.5.2.2 Physico-chemical changes

Disturbance caused by the physical process of macrophyte removal may also be partly responsible for the results observed in this study. Mechanical macrophyte removal causes physical and chemical changes in the water column that can negatively impact the resident fauna (Brookes,

1988; Hudson and Harding, 2004). Macrophyte removal has been reported to temporarily increase suspended sediment concentration (Hudson and Harding, 2004; Young et al., 2004) which interferes with normal respiration in fish by reducing oxygen availability (Waterman et al., 2011) and clogging the gills (Bruton, 1985), and limits the feeding success of predatory species by increasing turbidity and invertebrate drift (Ryan, 1991; Quinn et al., 1992; Wood and Armitage, 1997; Hazelton and Grossman, 2009a). Although these factors were not measured in the current study due to logistical constraints, high levels of suspended sediment were observed during experimental macrophyte removal, and it is possible that this led to the short term decreases in CPUE observed in the treated reaches. From the physico-chemical variables examined, only water temperature changed significantly in both treatment groups, but this was unlikely to explain the observed changes in the fish community. Decreases in temperature were probably the result of seasonal changes in climate, and were unlikely to be associated with macrophyte removal. Temperatures in the treatment groups and the control were relatively similar after excavation, and the larger decreases in temperature seen in the cleared and staggered areas were a reflection of the relatively low temperatures recorded in the undisturbed reaches prior to macrophyte removal. Determining the physico-chemical drivers behind changes in the fish community following macrophyte removal is vital if the impacts of this activity are to be effectively managed. Chapters 3 and 5 will address how large scale macrophyte removal impacts sediment resuspension and dissolved oxygen in New Zealand waterways. Chapter 4 will examine how sediment resuspension during macrophyte removal impacts the feeding and respiratory abilities of resident fish.

Chapter 3.0: The effects of mechanical macrophyte control on suspended sediment concentrations in southern New Zealand streams

3.1 Abstract

Suspended sediment (SS) is an important from of pollution in aquatic ecosystems, and can be detrimental to freshwater fish. Although macrophytes mediate sediment deposition, little effort has been put into determining how their removal affects sediment resuspension. The present study examined the immediate impacts of small-spatial-scale experimental excavation of macrophytes, and the long-term impacts of a large-spatial-scale macrophyte removal operation on SS concentrations. To determine the effects of small-spatial-scale experimental macrophyte removal on sediment resuspension, SS concentrations were monitored during mechanical excavation of 350-metre sections of three streams in the catchment of Waituna lagoon in New Zealand's South Island. To determine the long-term effects of large-spatial-scale macrophyte removal on sediment resuspension, SS concentrations were monitored at eight locations in the same catchment before, during and after a macrophyte removal operation in which over 80 km of waterway was excavated. The results of this study suggest that bed disturbance during mechanical excavation of macrophytes significantly increases SS concentration in the short term. Significant long-term increases in SS were also observed, indicating that without macrophytes to encourage the deposition and retention of SS after mechanical excavation, disturbed material is continually resuspended by fluvial processes. The results demonstrate that increased SS following macrophyte removal may be an important driver of reduced fish abundance, and indicate that this activity may be more detrimental to native fish than previously thought.

3.2 Introduction

3.2.1 Importance of suspended sediment (SS) to freshwater fishes

Anthropogenic activities that increase sediment input can be detrimental to freshwater fish populations (Alabaster and Lloyd, 1982; Lazar et al., 2010; Kemp et al., 2011). A commonly cited effect of increased SS on post-hatching life-stages of fish is reduced respiratory performance (Bruton, 1985; Henley et al., 2000; Bilotta and Brazier, 2008; Kemp et al., 2011). Oxygen consumption by anoxic SS can significantly reduce environmental dissolved oxygen (DO) during resuspension (DiToro, 2001; Waterman et al., 2011; Krevs and Kucinskiene, 2012). The amount of oxygen a fish can absorb across fine gill membranes is dependent on external oxygen conditions, and reductions in environmental DO are reflected in the supply of oxygen to body tissues (Dean and Richardson, 1999). In addition, suspended particulates bind to and abrade the delicate gill structures of fish. This reduces the surface area of the gills available for gaseous exchange, further restricting oxygen uptake (Lake and Hinch, 1999; Sutherland and Meyer, 2007).

Reduced invertebrate abundance following sediment resuspension hinders the ability of some fish species to meet metabolic energy demands, leading to decreased growth rates (Quinn et al., 1992; Wood and Armitage, 1999; Kemp et al., 2011). For visual foragers, the ability to locate and obtain food resources is also impaired during sediment resuspension. Increased turbidity/reduced visibility decreases the reactive distance to prey items, reduces capture success and increases energy expenditure during feeding (Sigler et al., 1984; Sutherland and Meyer, 2007; Hazelton and Grossman, 2009a; Kemp et al., 2011). The physiological stress of reduced feeding and respiratory performance can increase fish mortality and migration during periods of elevated SS, leading to shifts in community structure (Redding et al., 1987; Boubée et al., 1997; Brown et al., 1998; Lake and Hinch, 1999; Robertson et al., 2007; Crosa et al., 2010; Kemp et al., 2011).

Sensitivity to SS is dependent on species and life stage, and differs markedly between New Zealand fishes. Boubée et al. (1997) determined that the turbidity needed to elicit a 50 % avoidance response to SS ranged from 25 to > 1,100 nephelometric turbidity units (NTU) between juvenile-stage banded kokopu (*Galaxias fasciatus;* 25 NTU), koaro (*Galaxias brevipinnis;* 70 NTU), inanga (*Galaxias maculatus;* 420 NTU), redfinned bully (*Gobiomorphus huttoni;* > 1,100 NTU), shortfin eel (*Anguilla australis;* > 1,100 NTU) and longfin eel (*Anguilla dieffenbachii;* > 1,100 NTU). The drivers behind avoidance are unclear, but reduced feeding ability in turbid environments (> 160 NTU) has been observed in juvenile banded kokopu, smelt (*Retropinna retropinna*), inanga and common bully (*Gobiomorphus cotidianus*) (Rowe and Dean, 1998). The

response of adult fish populations to increased SS has not been extensively studied in New Zealand, but the regular occurrence of SS concentrations over 120 milligrams per litre (mg L^{-1}) can reduce the occurrence of sensitive species such as banded kokopu and redfin bullies (Rowe et al., 2000). Therefore, activities that increase sediment load may have a negative impact on native fish communities (Richardson and Jowett, 2002). Macrophytes are key drivers of sediment deposition and retention in lowland streams, yet little effort has been put into determining how their removal impacts SS.

3.2.2 Macrophytes and sediment

The potential for sediment resuspension and associated changes in water chemistry during bed disturbance is greatest in streams where eutrophication has allowed macrophytes to proliferate. The increased flow resistance within macrophyte stands reduces near-bed velocity and turbulence, increasing the deposition and retention of fine particulates that would otherwise erode (Luhar et al., 2008; Jones et al., 2012). Accumulation of these particulates limits oxygen transfer between the water and stream bed to the top two to five millimetres of sediment, promoting anoxic conditions in the layers below (Simpson et al., 1998). Below this depth, anaerobic microbial decomposition mineralises and reduces particulate organic matter to soluble intermediates (DiToro, 2001). When the sediment is resuspended, these intermediates are rapidly consumed through bacterial metabolism or chemical oxidization, both of which reduce DO in the water column (DiToro, 2001; Waterman et al., 2011; Krevs and Kucinskiene, 2012).

3.2.3 Mechanical excavation of macrophytes and SS

Although mechanical excavation of macrophytes represents a significant source of bed disturbance (Hudson and Harding, 2004), the effects on sediment resuspension are unclear. Increased deposition of sediment has been observed downstream of excavation works in English channels, but only under base-flow conditions (Brookes, 1988). To date, increases in SS observed following experimental excavation in New Zealand streams have been short-lived. In the study presented in Wilcock et al. (1998) turbidity remained elevated for less than four hours following excavation of 80 metres of a Waikato waterway. Similarly turbidity recovered rapidly after excavation of an isolated section of a Marlborough drain (Young et al., 2004). Because increased sediment resuspension following excavation has been short-lived in these studies, it has been suggested that SS has little impact on freshwater communities after excavation (Young et al., 2004). Given the small spatial scale of past studies where experimental excavation has been limited to short isolated sections of waterway (Wilcock et al., 1998; Hudson and Harding, 2004; Young et al., 2004), it is

possible that the long-term effects of macrophyte removal on SS concentration have been underestimated. Drain maintenance is often carried out on a much larger temporal and spatial scale, with entire waterways or catchments being excavated over a number of weeks. In this situation SS concentrations in some areas of the lower catchment are likely to remain elevated for longer than reported in Wilcock et al. (1998) and Young et al. (2004).

Sediment suspended during macrophyte removal may also have significant impacts on fish communities when deposited in downstream receiving environments. Deposited fine sediment can alter fish growth and community structure (Henley et al., 2000; Kemp et al., 2011) by smothering developing eggs (Kemp et al., 2011), reducing oxygen availability near the benthos (Bruton, 1985; Henley et al., 2000), altering habitat suitability and availability (Berkman and Rabeni, 1987; Walling and Amos, 1999; Yamada and Nakamura, 2002; Collins and Walling, 2007) and reducing the availability of invertebrate prey (Wood and Armitage, 1999; Matthaei et al., 2006). Time and budgetary constraints meant that the effects of macrophyte removal on sediment deposition and fish communities in downstream receiving environments were not quantified in this thesis. Suspended sediment was judged to pose a more immediate and quantifiable threat to fish after macrophyte removal and, consequently, became the focus of this study.

3.2.4 Aims

The aims of this study were to determine how mechanical macrophyte removal influences sediment regimes in lowland streams. Specifically, changes in the relationship between SS and flow were quantified during small and large-spatial-scale macrophyte removal operations, and the time these effects persisted was measured. This study is the first to measure long-term changes in sediment-flow relationships following a large-spatial-scale macrophyte removal operation. It was hypothesised that bed disturbance during mechanical excavation of macrophytes would increase SS concentrations. Because macrophytes provide bed stability, sediment disturbed during excavation is likely to be continually resuspended until it is transported out of the system, or emerging macrophytes allow it to settle. Therefore, SS concentrations were predicted to remain elevated for an extended period after large-spatial-scale macrophyte removal, particularly during periods of high flow.

3.3 Methods

3.3.1 Study area

This study was carried out in the catchment of Waituna Lagoon in the Southland Region of New Zealand's South Island. The lagoon's drainage basin encompasses approximately 20,000 hectares of high conservation value wetland, intensively developed farmland and native forest. Of the three creeks that flow into the lagoon, the Waituna Creek has the largest catchment area (12,500 ha), followed by Carran Creek (5,700 ha) and Moffat Creek (1,700 ha) (Johnson and Partridge, 1998). The majority of the streams in the catchment have been extensively modified (Stevens and Robertson, 2007), and macrophytes are regularly excavated to ensure that large areas of reclaimed wetland within the catchment are drained effectively. Despite intensive agricultural development in recent decades, the waterways in the area provide important habitat for fish (Riddell et al., 1988). Native species present include banded kokopu, inanga, common smelt, common and redfinned bully and both shortfin and longfin eels (Riddell et al., 1988; Atkinson, 2008). The catchment also has a healthy population of giant kokopu (*Galaxias argenteus*), a large species of New Zealand galaxiid that is threatened in most other areas (Riddell et al., 1988; Allibone et al., 2010).

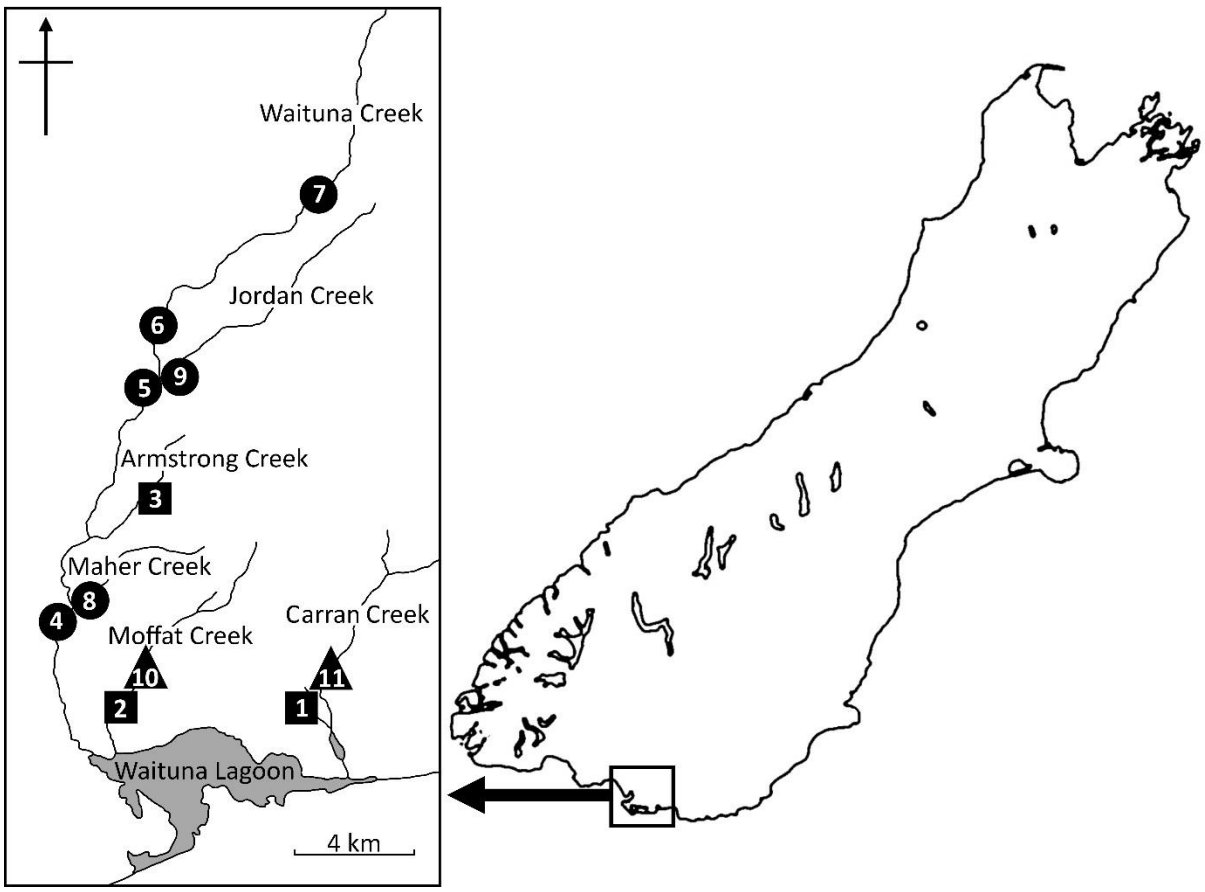

Figure 3.1 Study sites. Squares represent sites in which small-spatial-scale experimental excavation was conducted in 2011 (three sites). Circles represent sites excavated during the large-spatial-scale drainage operation conducted in 2012 (six sites) and triangles represent the control sites monitored during this operation (two sites). Numbers correspond to 1: Carran 1. 2: Moffat 1, 3: Armstrong, 4: Waituna 1, 5: Waituna 2, 6: Waituna 3, 7: Waituna 4, 8: Maher, 9: Jordan, 10:

3.3.2 Small-spatial-scale mechanical excavation

3.3.2.1 Study sites

The immediate impacts of mechanical excavation of macrophytes on SS concentration were examined at three 350-metre (m) study sites, each located in a different stream; Armstrong Creek, Moffat Creek and a tributary of Carrans Creek (sites hereafter referred to as Armstrong, Moffat 1 and Carran 1) (Figure 3.1). All study sites had > 50 % macrophyte coverage over the wetted width of the channel, had an average wetted width of between one and three meters, and had an average depth of between 20 and 35 centimetres.

3.3.2.2 Mechanical excavation

Macrophyte removal was carried out between the 22/03/2011 and the 25/03/2011 using a mechanical excavator, which worked from downstream to upstream. The excavator was equipped with a perforated bucket that removed both plant material and silt from the streambed, while allowing water to flush back into the channel (Figure 3.2). Material removed from the waterway was placed on the bank immediately adjacent to the channel.

Figure 3.2 Photograph of the two attachments used while excavating the study sites. a) Depicts the perforated bucket used to excavate silt and plant material from Armstrong, Carran 1 and Moffat 1 during experimental excavation in 2011. b) Depicts the weed rake used to excavate plant material from the remaining sites in 2012.

3.3.2.3 Water sampling and analysis

Water samples were collected at two locations during excavation of the treatment sites, 5 m below the most upstream point of the site and 50 m below the most downstream point. Five water samples were taken at 30-minute intervals at each location, with the first water sample being taken 5-10 minutes before excavation began.

Water sampling was carried out in accordance with the techniques described in McKenzie and Thomsen (1995). Sample jars were used to collect a one-litre water sample from the vertical centre of the water column. Samples were refrigerated or kept on ice before being analysed. Analysis was conducted by Hills Laboratories in Hamilton New Zealand, which measured SS concentration using techniques described in part 2540 D of Eaton et al. (1995). Volatile suspended solids (VSS) concentration was measured in each water sample using techniques described in part 2540 E of Eaton et al. (1995). VSS concentration provides an estimate of the amount of organic matter present in the suspended material.

3.3.2.4 Data analysis

Patterns in sediment resuspension during excavation were analysed descriptively. For each site, SS concentration, VSS concentration, and the proportion of SS comprised of VSS at both sampling locations were plotted against time.

3.3.3 Large-spatial-scale mechanical excavation

3.3.3.1 Mechanical excavation

Suspended sediment was monitored during a large macrophyte removal operation conducted by Environment Southland, the local government agency responsible for the maintenance of waterways in the lower South Island of New Zealand. The operation was conducted between the 08/02/2012 and the 28/03/2012. During this period two mechanical excavators cleared over 80 kilometres (km) of waterway along the main stem of the Waituna Creek and its tributaries. Excavation began in the lower reaches of the catchment and moved upstream. Excavators were equipped with weed rakes designed to remove vegetation from the waterway without extracting silt (Figure 3.2). Material removed from the waterway was placed on the bank immediately adjacent to the channel.

3.3.3.2 Study sites

Eight sampling sites in the catchment of Waituna Lagoon were selected from existing water-quality monitoring programmes run by Environment Southland. Six were within the area targeted for excavation. These were located on the main stem of Waituna Creek (four sites hereafter referred to as Waituna 1, Waituna 2, Waituna 3 and Waituna 4) and two of its larger tributaries, Jordan Creek and Maher Creek (two sites, hereafter referred to as Jordan and Maher) (Figure 3.1). The remaining two sites were located on Moffat Creek and Carran Creek, and provided a "control" as neither stream was extensively cleared during the study period (sites hereafter referred to as Moffat 2 and Carran 2) (Figure 3.1).

3.3.3.3 Data sources

Data included in the analyses came from three sources, a long-term water quality data collected by Environment Southland, regular water sampling carried out over the macrophyte removal period (present study) and continual turbidity and flow measurements taken over the same period (present study).

3.3.3.3.1 Long term water quality and flow data

Long term records of SS concentrations, based on monitoring conducted by Environment Southland since 2001 or longer, exist for Waituna 1, Waituna 3, Moffat 2 and Carran 2. In total, 789 water samples were taken from the four study sites between 1995 and January 2012 (Table 3.1). The variables analysed from samples have changed through time; SS concentration was only measured in 331 samples (Table 3.1).

Discharge data, recorded in cubic metres per second ($m^3 s^{-1}$), at the time of collection are available for all samples from which SS concentrations were measured between August 2001 and January 2012 (Table 3.1). At Waituna 1, Moffat 2 and Carran 2 flow rates were derived from real-time water level data applied to stage-discharge rating curves developed by spot gauging. Between 2001 and 2007, and from April 2011, water level was measured at 10-minute intervals at Waituna 1 by means of a Vegapuls WL61 radar. From April 2011, water levels at Moffat 2 and Carran 2 were collected using Trutrack capacitance probes that logged data at 10-minute intervals.

All flow data from Waituna 3, and flow data from Moffat 2 and Carran 2 prior to the installation of the Trutrack capacitance probes (April 2011) were calculated from the water level at the nearby Waihopai River. A Vegapuls WL61 radar has been in place in this river since 2001, allowing flow

to be calculated every ten minutes from a stage-discharge rating curve. Manual flow-gauging demonstrated that flows at the three study sites are strongly correlated with flow at the river (Waituna 3 $R^2 = 0.9828$; Moffat 2 $R^2 = 0.969l$; Carran 2 $R^2 = 0.926$) and can be simulated from the continual water level record at Waihopai (C. Jenkins and G. Scott, Environment Southland, pers. comm.)

3.3.3.3.2 Water sampling and flow measurements during excavation

Collection of water samples for water quality analyses began on the 24/01/2012, 15 days before macrophyte removal and, with the exception of Waituna 1, continued until the 03/05/2012, 45 days after drainage operations were completed. The time between consecutive samples generally ranged from one to four days, but increased after excavation was completed. In Waituna 1, water sampling continued until the 08/08/2013, but from May 2012, collection was reduced to an average of 2 times a month. In total, 403 samples were taken during the sampling period (Table 3.1).

Discharge data ($m^3 s^{-1}$) at the time of collection are available for 288 of the samples collected from the eight study sites between January 2012 and August 2013 (Table 3.1). Flows in Waituna 2, Waituna 4, Maher and Jordan were calculated by manual gauging. At each gauging a transect was placed at the same point across the channel. Wetted width along the transect was recorded, and water depth measured at 15 to 20 points along the transect. Water velocity was measured at each point using a SonTek FlowTracker ADV. When water depth was < 0.5 m, velocity was measured once at 60 % of water column height. If depth was > 0.5 m, velocity was measured at 20 % and 80 % of water column height. Depth and width measurements were used to calculate cross-sectional area. This was then multiplied by mean water velocity to provide a measure of flow rate.

At Waituna 1, Moffat 2 and Carran 2 flows were derived from real-time water level data applied to stage-discharge rating curves developed from spot gaugings. Water level was monitored at 10 minute intervals at the sites using a Vegapuls WL61 radar at Waituna 1 and Trutrack capacitance probes at Moffat 2 and Carran 2. As with the long-term data record, flow at the Waituna 3 was simulated from the water level record at Waihopai River.

3.3.3.3.3 Turbidity monitoring

Between the 03/02/2012 and the 15/05/2012, a Greenspan TS 300 Turbidity Probe recorded turbidity at 10 minute intervals at Waituna 1. Turbidity measurements were accurate up to 1,000 NTU. Values recorded by the probe exceeded this level on a number of occasions, and only data collected between the 08/02/2012 and the 04/04/2012 are considered reliable. SS concentrations

measured from water samples were regressed against the turbidity measurements recorded at the time of sample collection. The following regression equation, was used to calibrate turbidity data to SS concentrations.

$$Sediment\ concentration = Turbidity \times 0.8714$$

Despite the strong relationship between turbidity and SS concentration (R^2 = 0.862 P < 0.001), the regression analysis only included turbidity values up to 600 NTU. Above this level, SS concentrations calculated from this relationship are extrapolations.

Between the 07/02/2012 and the 04/04/2012, a second Greenspan TS 300 Turbidity Probe recorded turbidity at five minute intervals at Waituna 3. Unlike at Waituna 1 turbidity measurements at Waituna 3 were only accurate up to 400 NTU. The decision not to use a turbidity probe with a greater measurement range (0-1000 NTU) was based on the availability of equipment and the expectation that turbidity would be lower at this site than at the downstream Waituna 1 site. SS concentrations measured from water samples were regressed against the turbidity measurements recorded at the time of sample collection. Turbidity data was then calibrated to SS concentration based on equation of the regression line:

$$Sediment\ concentration = Turbidity \times 1.7592$$

Despite the strong relationship between turbidity and SS concentrations (R^2 = 0.811 P < 0.001), the regression analysis only included turbidity values up to 73 NTU. SS concentrations calculated from higher values are extrapolations.

Table 3.1 The number of water samples, the number of SS measurements, and the number of corresponding flow measurements taken at each site.

	Long term water sampling		
Site	Samples	SS measured	Flow recorded
Waituna 1	273	70	70
Waituna 3	128	44	44
Moffat 2	193	107	107
Carran 2	195	110	110
	Water sampling during and after excavation		
Site	Samples	SS measured	Flow recorded
Waituna 1	79	79	75
Waituna 2	47	47	22
Waituna 3	48	48	38
Waituna 4	50	50	22
Jordan	45	45	22
Maher	47	47	22
Moffat 2	44	44	44
Carran 2	43	43	43
	Total		
Site	Samples	SS measured	Flow recorded
Waituna 1	352	149	145
Waituna 2	47	47	22
Waituna 3	176	92	82
Waituna 4	50	50	22
Jordan	45	45	22
Maher	47	47	22
Moffat 2	237	151	151
Carran 2	238	153	153
Total	1192	734	619

3.3.3.4 Data Analysis

3.3.3.4.1 Data exploration and transformations

Two separate data sets were compiled for each site. One comprised data collected before an excavator passed the site, and the other, data collected after this point. In the case of the two control sites, all data collected during and after the macrophyte control operations were treated as the "after" data, while the long-term water quality record were treated as the "before" data. The long-term water-quality data were added to the before data of Waituna 1, and Waituna 3 to create a robust before-after-control-impact (BACI) design (Stewart-Oaten et al., 1986). SS concentrations calculated from turbidity measurements taken at Waituna 1 after excavation were analysed independently from the after data collected from water samples.

The Kolmogorov-Smirnov test was used to test data for the assumption of normality before statistical analyses were conducted. $Log10_x$ and log_{x+1} transformations were used to normalise distributions when this assumption was not met. Parametric tests were used whenever possible, but when data could not be transformed to approximate normality, non-parametric analyses were conducted. Regression analyses were used to evaluate the influence of flow on SS concentration before and after excavation in each site (independent variable = flow; dependant variable = SS concentration). Flow was treated as a covariate in statistical comparisons of SS concentrations in sites where the results of regression analyses were significant ($P < 0.05$).

3.3.3.4.2 Parametric analyses

Analyses of covariance (ANCOVA), with flow as a covariate, were used to compare SS concentrations measured from water samples before and after mechanical excavation at Waituna 1, Waituna 3, Moffat 2 and Carran 2. Flow was used as a covariate because of the strong relationship between SS concentration and flow at these sites. Sediment data collected from water samples without corresponding flow measurements were excluded from these analyses. All ANCOVAs met the assumption of homogeneity of regression slopes (Waituna 1 covariate interaction, $F_{108} = 2.582$, $P = 0.11$; Waituna 3 covariate interaction, $F_{69} = 3.00$, $P = 0.09$; Moffat 2 covariate interaction, $F_{147} = 0.003$, $P = 0.97$; Carran 2 covariate interaction, $F_{145} = 0.303$, $P = 0.52$). For ease of interpretation, estimated marginal means were calculated in each ANCOVA. The estimated marginal means are the mean SS concentrations before and after excavation estimated at the same flow rate (mean flow rate over the entire study period).

To determine if changes in SS concentrations persisted for the entire study period at Waituna 1, two additional ANCOVAs with flow as a covariate were conducted using turbidity data from the site. The first ANCOVA was used to compare SS concentrations calculated from turbidity measurements taken after macrophyte removal at Waituna 1 with SS concentrations measured from water samples taken before mechanical excavation [assumption of homogeneity of regression slopes met (ANCOVA covariate interaction, $F_{6915} = 2.210$, $P = 0.137$)]. The second ANCOVA was used to compare SS concentrations calculated from turbidity measurements taken in the final seven days of the reliable data record at Waituna 1 (29/03/2012-04/04/2012) with sediment concentrations measured from water samples taken before mechanical excavation [assumption of homogeneity of regression slopes met (ANCOVA covariate interaction, $F_{1032} = 2.192$, $P = 0.139$)]. While these analyses could be confounded by the different approaches used to derive SS concentrations before and after excavation (direct measurements from water samples vs. estimates from turbidity measurements), it is unlikely given the strong relationship between SS concentration and turbidity at the site (regression analysis, $R^2 = 0.862$ $P < 0.001$). Nevertheless, to ensure the results of this analysis reflected long-term changes in sediment regime, descriptive analysis of the water sampling data was also conducted (see below). Estimated marginal means were calculated in both of these ANCOVAs.

3.3.3.4.3 Non-parametric analyses

Mann-Whitney U-tests were used to compare SS concentrations before and after mechanical excavation in the four sites where data could not be transformed to approximate normality (Waituna 2, Waituna 4, Jordan, and Maher). Flow was not found to act as a covariate influencing SS concentrations in any of these sites, and was excluded from the analyses.

3.3.3.4.4 Descriptive analyses

Data collected from the fixed turbidity loggers at Waituna 1 and Waituna 3 were used to determine the distance the excavator needed to move upstream of the sites before point source sediment pollution (sediment suspended by the excavators) could no longer be detected. Turbidity-derived sediment concentrations and flow data collected from the sites were split into 24 hour (hr) periods (08:00 hrs 08:00 hrs). In each 24-hour period, mean turbidity and flow between 08:00 hrs and 20:00 hrs (when the excavators were working) and 20:00 hrs and 08:00 hrs (when the excavators were not working) were calculated. Normalised mean values and the associated standard error were plotted, and visual comparisons used to identify when non-flow-related differences in night and day SS concentrations ceased. Normalisation adjusts variance to give a common scale across

SS concentration and flow rate. Normalised values were calculated using the mean and standard deviation of day time and night time daily means [(x - mean)/ standard deviation]. The distance between the excavator and the study site when non-flow-related differences in night and day SS concentrations ceased was used to make inferences about the importance of point source sediment pollution following mechanical excavation.

Variation between the observed (measured from 79 water samples) and expected SS concentrations (based on the sediment-flow relationship at the site before excavation) at Waituna 1 after excavation were used to determine the length of time SS concentrations remained elevated above pre-excavation levels at the site. Expected concentrations were subtracted from the observed SS concentrations, and the values plotted against time. Expected sediment concentrations were calculated from flow at time of sample collection and the relationship between sediment and flow before excavation ($R^2 = 0.51$)

$$Expected\ SS_{log10} = (0.4776 \times Flow_{log10}) + 0.9619$$

3.4 Results

3.4.1 The effects of small-spatial-scale mechanical excavation on ss

3.4.1.1 Carran 1

Following excavation, SS concentration increased from 15 mg L^{-1} to 1,060 mg L^{-1} at the bottom of the site (Figure 3.3a). Increases at the top of the site were greater. As the excavator moved past the upstream sampling location SS concentration increased from 13 mg L^{-1} to 15,700 mg L^{-1}. The difference between the upstream and downstream sampling locations most likely reflects the low flow conditions at the site, and the 50 m buffer between the excavated section of waterway and downstream sampling location.

Following excavation, VSS concentration increased from 7 mg L^{-1} to 490 mg L^{-1} at the bottom of the site (Figure 3.3b). VSS concentration at the top of the site increased from 7 mg L^{-1} to 6,800 mg L^{-1} once the excavator had moved past the upstream sampling location. The volatile proportion of suspended material remained relatively constant at both sampling locations, varying between 28.7 % and 53.8 % (Figure 3.3c).

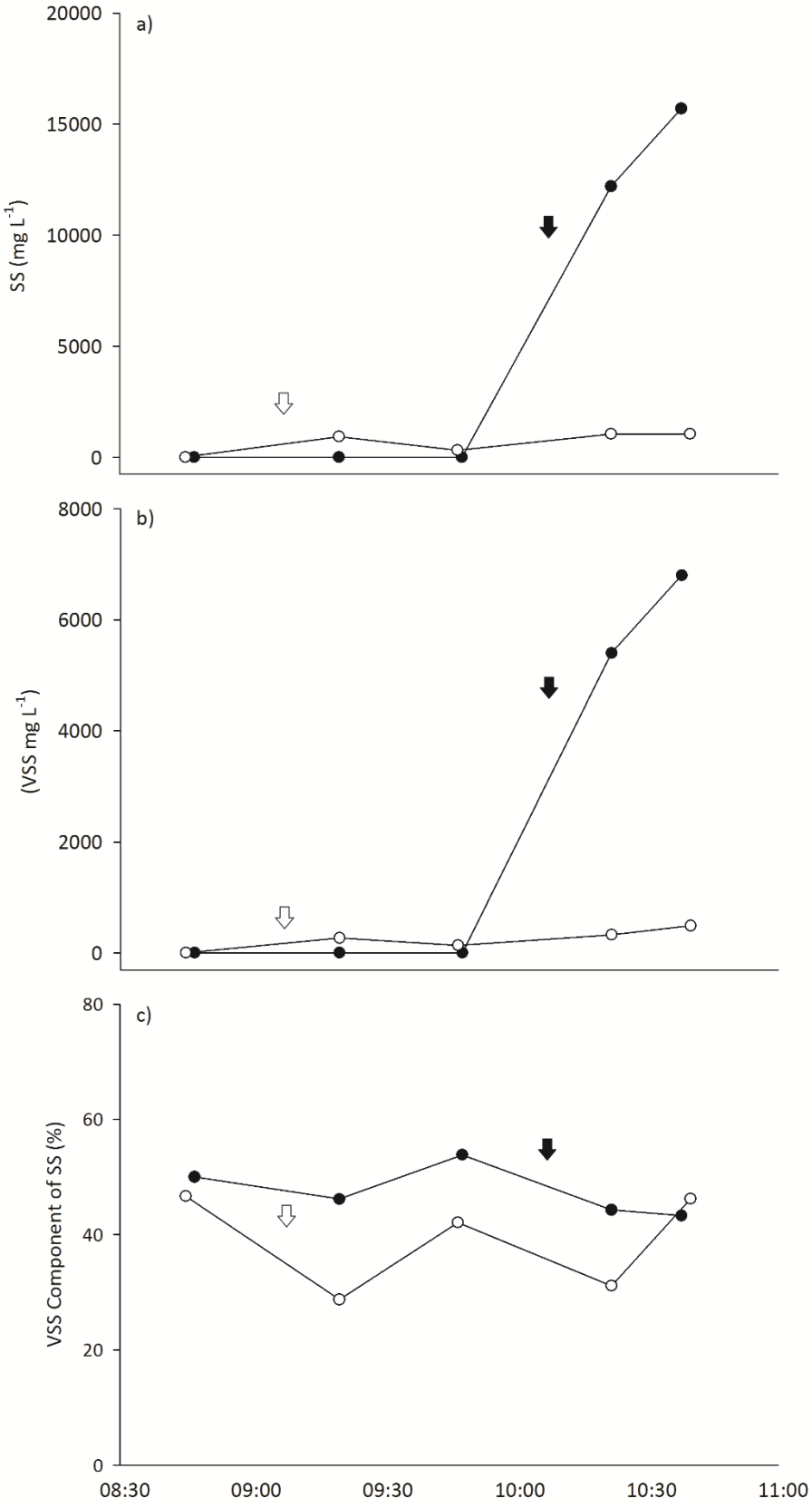

Figure 3.3 SS concentration (a), VSS concentration (b) and the percentage of SS comprised of VSS (c) during excavation of a 350-m section of Carran 1. Black circles represent concentrations at the upstream end of the site while white circles represent concentrations 50 m downstream of the site. Arrows represent when the excavator moved past the sampling locations.

3.4.1.2 Moffat 1

Following excavation, SS concentration increased from 6 mg L^{-1} to 630 mg L^{-1} at the bottom of the site. SS concentration at the top of the site increased from 5 mg L^{-1} to 2,300 mg L^{-1} once the excavator moved past the upstream sampling location (Figure 3.4a). Again the greater observed increase upstream compared to downstream most likely reflects the high residence time of water in the site, and the 50 m buffer between the excavated section of waterway and the downstream sampling location.

Following excavation VSS concentration increased from 3 mg L^{-1} to 95 mg L^{-1} at the bottom of the site. As the excavator moved past the upstream sampling location VSS concentration increased from 3 mg L^{-1} to 39 mg L^{-1} (Figure 3.4b). Accompanying the relatively minor increases in VSS concentration at the upstream sampling location was a marked decrease in the volatile proportion of suspended material. Prior to excavation, upstream VSS comprised 75 % of suspended material. This rapidly decreased to 1.6% once the excavator moved past the sampling location. The volatile proportion of suspended material also decreased from 50 % to 5.9 % at the downstream sampling location, but had increased to 23.9 % 120 minutes after excavation began (Figure 3.4c).

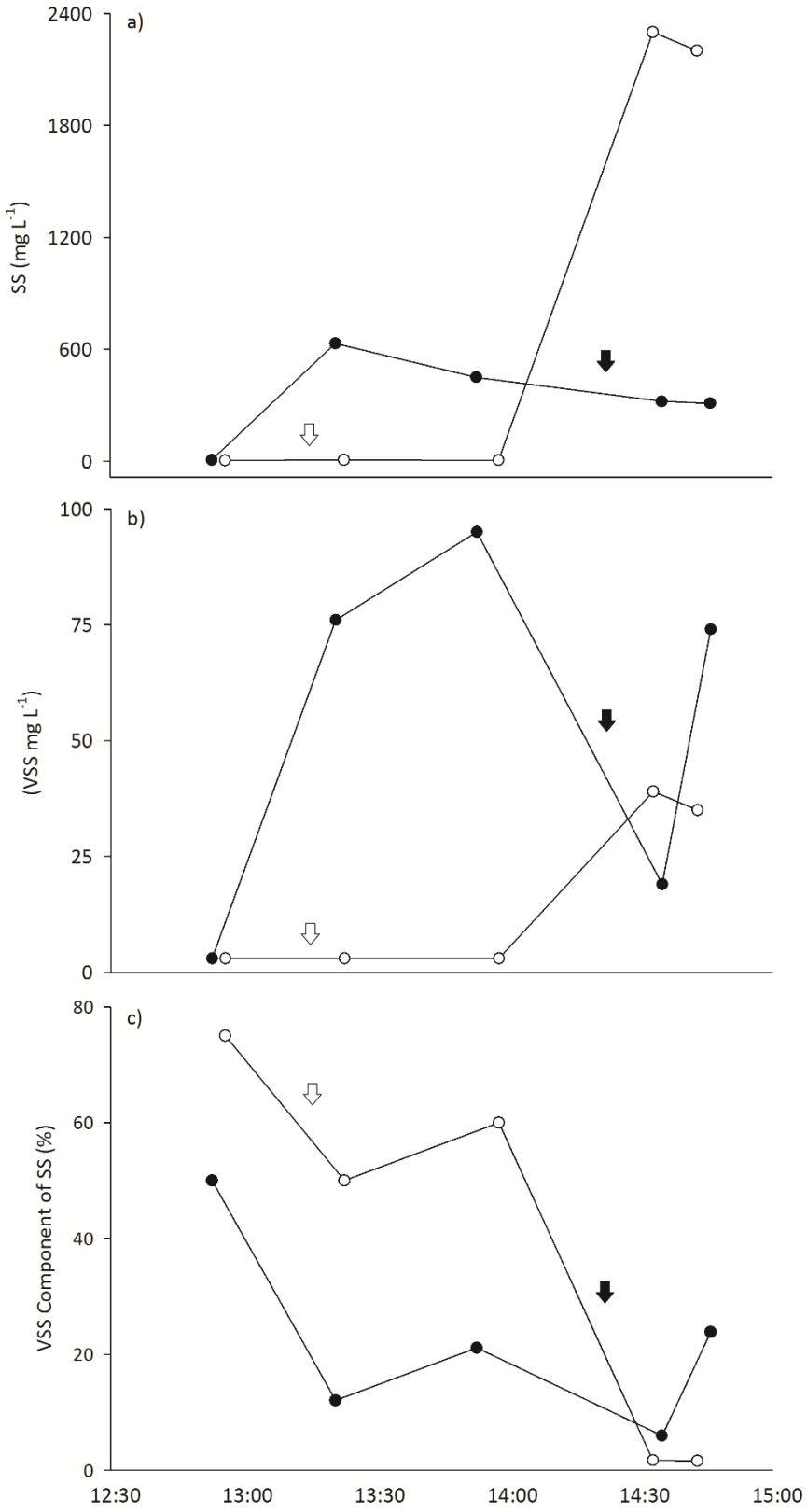

Figure 3.4 SS concentration (a), VSS concentration (b) and the percentage of SS comprised of VSS (c) during excavation of a 350-m section of Moffat 1. Black circles represent concentrations at the upstream end of the site while white circles represent concentrations 50 m downstream of the site. Arrows represent when the excavator moved past the sampling locations.

3.4.1.3 Armstrong

Following excavation SS concentration increased from 4 mg L^{-1} to 1,890 mg L^{-1} at the bottom of the site (Figure 3.5a). Although a steady decrease over the next 90 minutes was observed, SS concentration had not dropped below 1,200 mg L^{-1} 120 minutes after excavation began. The excavator failed to reach the top of the site in the sampling period, and SS concentration at the upstream sampling location remained at 4 mg L^{-1} over the entire excavation period.

VSS concentration increased from 3 mg L^{-1} to 700 mg L^{-1} at the bottom of the site (Figure 3.5b). VSS concentrations decreased over the next 90 minutes, but had not dropped below 400 mg L^{-1} 120 minutes after excavation began. Although VSS concentration increased following excavation, the proportion of SS comprised of VSS decreased markedly after excavation began. Prior to excavation VSS made up 75 % of suspended material, 30 minutes after excavation began this had decreased to 34.8 % (Figure 3.5c).

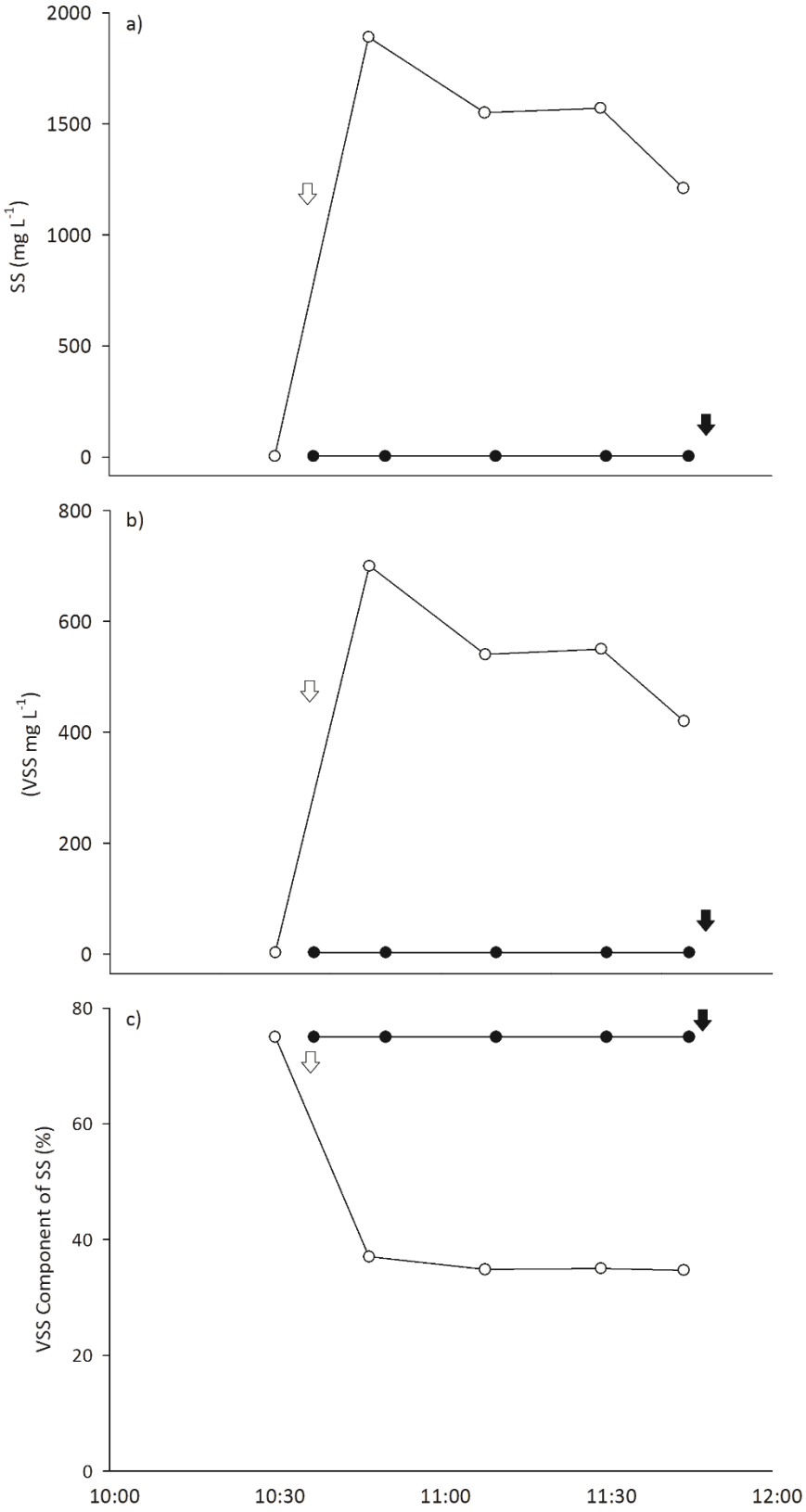

Figure 3.5 SS concentration (a), VSS concentration (b) and the percentage of SS comprised of VSS (c) during excavation of a 350-m section of Armstrong Creek. Black circles represent concentrations at the upstream end of the site while white circles represent concentrations 50 m downstream of the site. Arrows represent when the excavator moved past the sampling locations.

3.4.2 The effects of large-spatial-scale mechanical excavation on ss

3.4.2.1 SS concentrations measured from water samples

SS concentrations in water samples increased significantly at all treatment sites following mechanical excavation (Figure 3.6). Estimated marginal mean SS concentrations of water samples collected from Waituna 1 increased from 11.93 mg L^{-1} (±1.09) to 87.90 mg L^{-1} (±1.18) (calculated at 1.89 m^3 s^{-1} flow) (ANCOVA, $F_{109} = 92.663$, $P < 0.001$, partial eta-squared = 0.459) following excavation. At Waituna 3 estimated marginal mean SS concentration increased from 5.10 mg L^{-1} (±0.086) to 26.28 mg L^{-1} (±0.169) (calculated at 0.29 m^3 s^{-1} flow) (ANCOVA, $F_{70} = 70.608$, $P < 0.001$, partial eta-squared = 0.459). Mean SS concentration increased from 3.33 mg L^{-1} (±0.19) to 116.05 mg L^{-1} (±55.45) at Waituna 2 (Mann-Whitney U-test, $P < 0.001$), from 3.51 mg L^{-1} (±0.18) to 28.7 mg L^{-1} (±21.32) at Waituna 4 (Mann-Whitney U-test, $P = 0.004$), from 3.08 mg L^{-1} (±0.55) to 111.0 mg L^{-1} (±54.7) at Jordan (Mann-Whitney U-test, $P < 0.001$) and from 6.8 mg L^{-1} (±0.83) to 209.29 mg L^{-1} (±111.97) at Maher (Mann-Whitney U-test, $P < 0.001$). Conversely SS concentrations in the two control sites did not differ significantly between the macrophyte removal period and the long-term water quality record (ANCOVA, Moffat 2 $F_{148} = 1.1078$, $P = 0.301$, partial eta-squared = 0.007; Carran 2 $F_{146} = 0.354$, $P = 0.553$ partial eta-squared = 0.002).

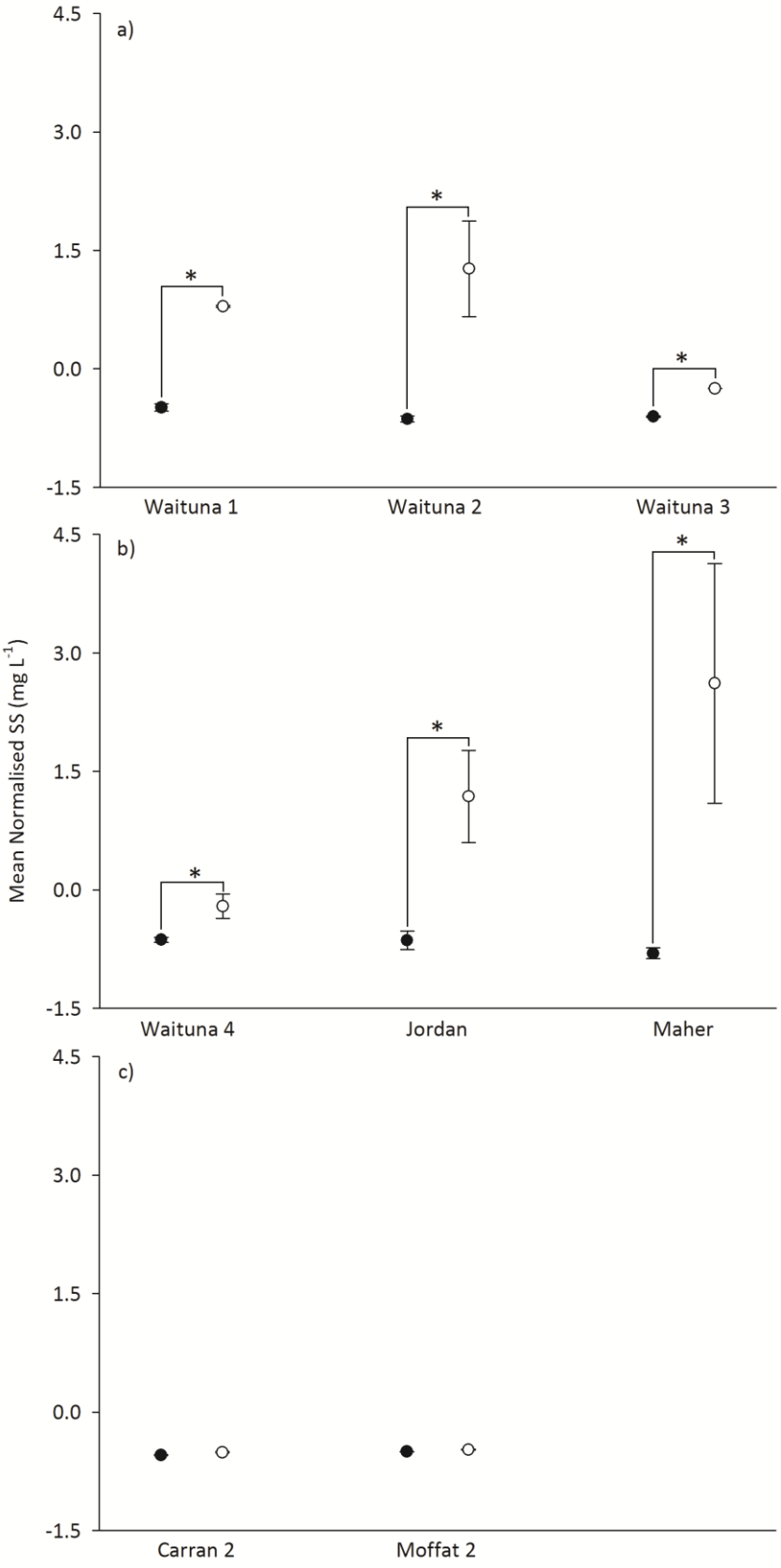

Figure 3.6 Mean normalised SS concentrations (± SE) recorded in excavated sites (a-b) and control sites (c) before (black circles) and after (white circles) excavation. Estimated marginal means are presented for Waituna 1, Waituna 3, Moffat 2 and Carran 2. All other values presented are arithmetic means. Statistically significant differences are illustrated with an * ($P < 0.05$).

3.4.2.2 SS concentrations derived from turbidity data

SS concentrations at Waituna 1 calculated from turbidity data collected after excavation were significantly greater than those recorded from water samples taken before macrophyte removal (ANCOVA, F_{6916} = 908.07, P < 0.001, partial eta-squared = 0.116) (Figure 3.7a). Estimated marginal mean SS concentrations increased from 5.09 mg L^{-1} (±0.006) to 50.52 mg L^{-1} (±0.07) (calculated at 0.55 m^3 s^{-1} flow) after macrophyte removal. Sediment concentrations remained elevated at this site throughout the entire study period. SS concentrations calculated from turbidity data collected 44 to 50 days after excavation were significantly greater than those recorded from water samples taken before macrophyte removal (ANCOVA, F_{1033} = 452.357, P < 0.001, partial eta-squared = 0.334) (Figure 3.7b). The estimated marginal mean SS concentration over this period was 30.477 mg L^{-1} (±0.013) compared to 5.367 mg L^{-1} (±0.071) before excavation (calculated at 0.41 m^3 s^{-1} flow) (Table 3.2).

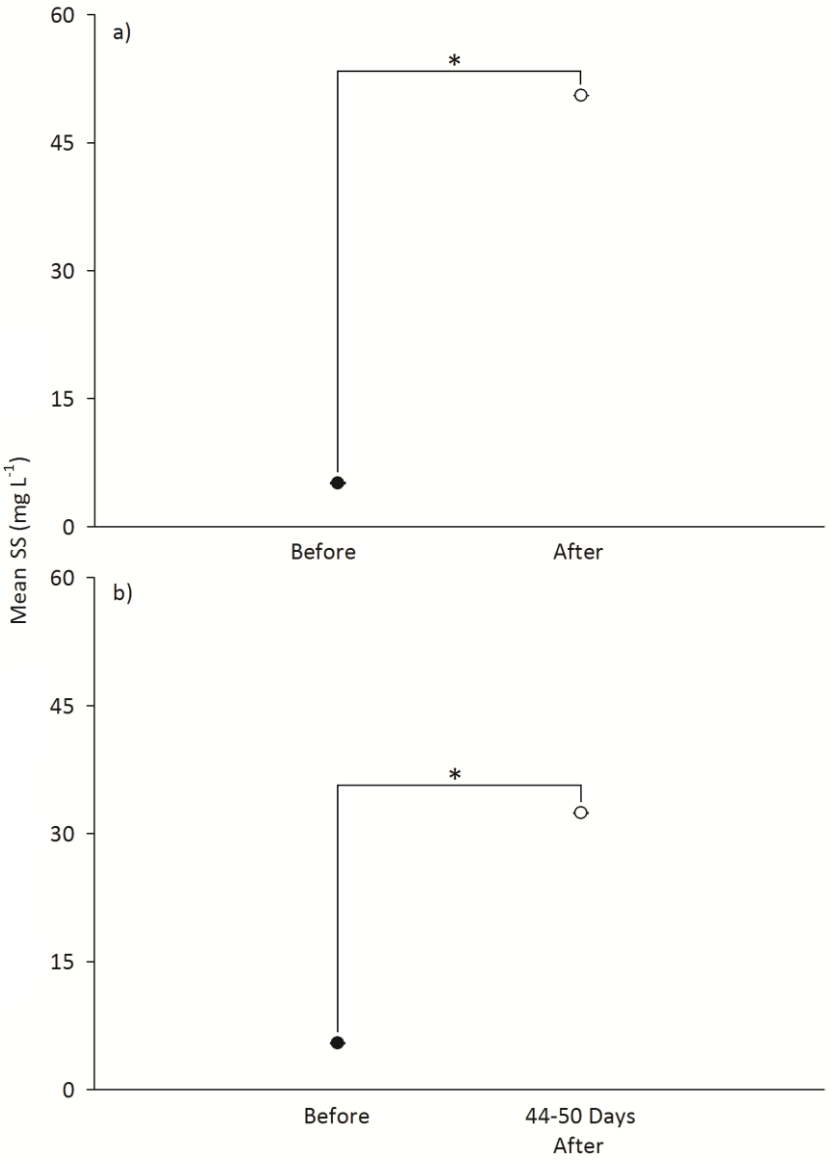

Figure 3.7 a = Mean SS concentrations (±SE) recorded at Waituna 1 before (measured from water samples) and after excavation (calculated from turbidity data). The values presented are estimated marginal means calculated at 0.55 m^3 s^{-1} flow. b = Mean SS concentrations (±SE) calculated from turbidity data recorded at Waituna 1 before (measured from water samples) and 44-50 days after excavation (calculated from turbidity data). The values presented are estimated marginal means calculated at 0.41 m^3 s^{-1} flow. Statistically significant differences in SS concentration before and after excavation are illustrated with an * ($P < 0.05$).

3.4.2.3 Pre and post excavation SS maxima

There was a marked difference in the maximum SS concentrations recorded before and after excavation in the treatment sites. The highest concentration of SS was recorded in the Maher site immediately after excavation. Before treatment, the maximum SS concentration recorded at this site was 11 mg L^{-1}. On the day the site was excavated (09/02/2012) this increased to 3700 mg L^{-1}. Post-excavation maxima in the other treatment sites were recorded during a period of high flow on the 15/03/2012. At Waituna 1, the post-excavation SS maximum of 630 mg L^{-1} was recorded on the 15/03/2012 during flows of 4.791 m^3 s^{-1}. This exceeded the pre-excavation maximum of 96 mg L^{-1} which was recorded during flows of 19.22 m^3 s^{-1}. At Waituna 3, the maximum post-excavation SS concentration of 540 mg L^{-1} was recorded at 1.48 m^3 s^{-1} of flow on the 15/03/2012. The maximum pre-excavation concentration at this site was 97 mg L^{-1}, and was measured during comparable flow conditions (1.27 m^3 s^{-1}). Pre-excavation maxima did not exceed 10 mg L^{-1} in Waituna 2, Waituna 4 or Jordan. Maximum post-excavation SS concentrations at all three of sites were recorded on the 15/03/2012, and were 860 mg L^{-1} at Waituna 2, 220 mg L^{-1} at Waituna 4 and 960 mg L^{-1} at Jordan.

High-flow events during the macrophyte removal period did not result in comparable increases in sediment resuspension in the two control sites. The maximum SS concentrations at Moffat 2 and Carran 2 during the drain-clearing period were recorded during high flows on the 15/03/2012 (Moffat 2 = 57 mg L^{-1} at 0.91 m^3 s^{-1} flow; Carran 2 = 40 mg L^{-1} at 1.17 m^3 s^{-1} flow), but did not exceed maximum concentrations observed before macrophyte removal at the treatment sites (Moffat 2 = 133 mg L^{-1} at 2.48 m^3 s^{-1} flow; Carran 2 = 100 mg L^{-1} at 2.54 m^3 s^{-1} flow).

Table 3.2 Maximum SS concentrations recorded before and after macrophyte control.

	Excavated					
	Before			After		
Site	Max SS (mg L^{-1})	Date	Flow (m^3 s^{-1})	Max SS (mg L^{-1})	Date	Flow (m^3 s^{-1})`
Waituna 1	96	16/05/11	19.22	630	15/03/12	4.79
Waituna 2	8	01/02/12	0.05	860	15/03/12	-
Waituna 3	97	12/09/11	1.27	540	15/03/12	1.48
Waituna 4	7	07/02/12	0.006	220	15/03/12	0.26
Jordan	4	18/02/12	-	960	15/03/12	-
Maher	11	07/02/12	0.012	3700	08/02/12	-
	Control					
	Before			After		
Site	Max SS (mg L^{-1})	Date	Flow (m^3 s^{-1})	Max SS (mg L^{-1})	Date	Flow (m^3 s^{-1})`
Moffat	133	16/05/11	2.48	57	15/03/12	0.91
Carran	100	11/05/09	2.54	40	15/03/12	1.17

3.4.2.4 Patterns of sediment resuspension following excavation

Drainage works began in the Maher tributary on the 08/02/2012, approximately 1 km upstream of the Waituna 1 turbidity probe. The site was excavated on the 14/02/2012. Differences between day time and night time SS concentrations reflect the activity patterns of the excavators, which only worked during the day (Figure 3.8). Between the 12/02/2012 and the 17/02/2012 there was a marked difference between day time and night time SS concentrations, despite minimal differences in day time and night time flows during this period (Figure 3.8). After the 17/02/2012 differences in day time and night time SS concentrations were comparable to those observed before excavation. At this time, the closest excavator was approximately 3.2 km upstream of the site.

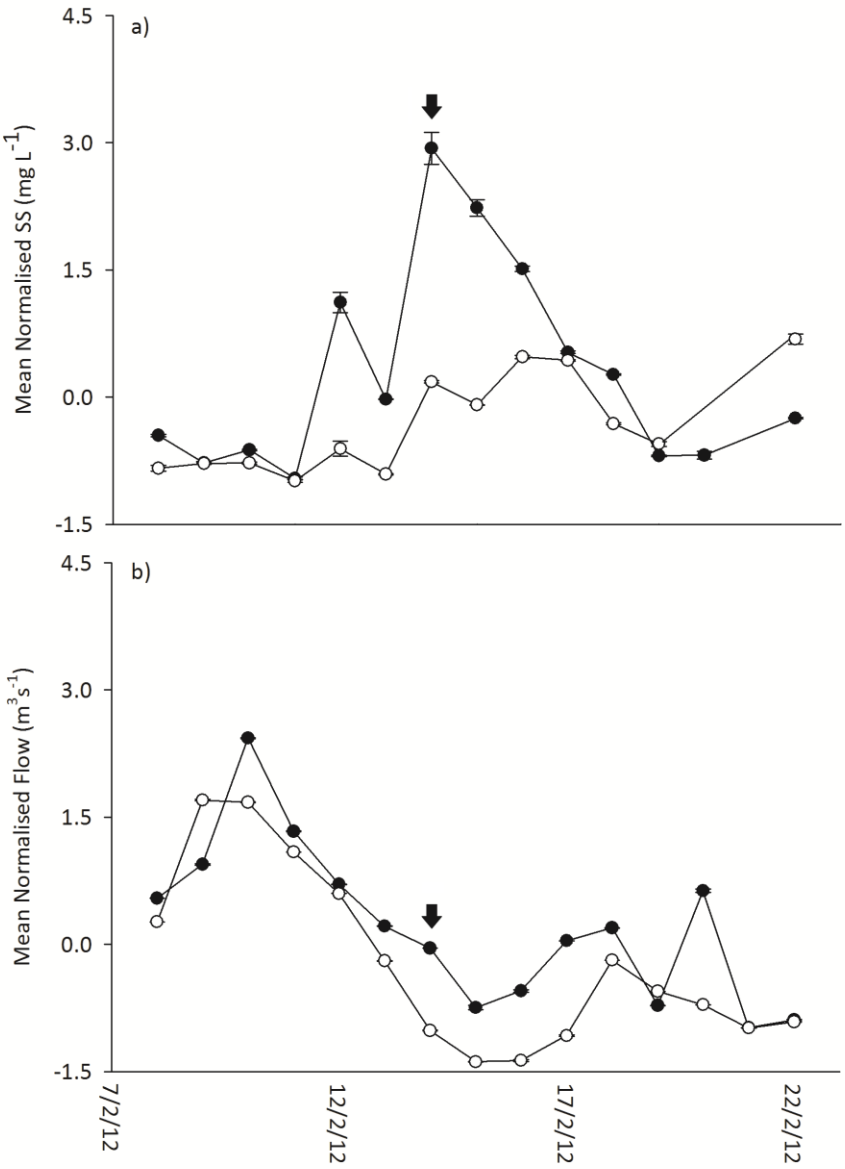

Figure 3.8 a = Daily mean SS concentration (± SE) recorded at Waituna 1 between 08:00 hrs and 20:00 hrs when the excavators were operating (black circles) and between 20:00 hrs to 08:00 hrs when no excavation was performed (white circles). b = Daily mean flow rate (± SE) recorded at Waituna 1 between 08:00 hrs and 20:00 hrs (black circles) and between 20:00 hrs and 08:00 hours (white circles). Arrows represent the date the excavator moved past the site.

The Waituna 3 site was excavated on the 08/03/2012. Between that date and the 11/03/2012 differences in day time and night time SS concentrations reflected the activities of the excavators working upstream of the site. During this period, day time SS concentrations were noticeably higher than those recorded at night, despite similar flow conditions on the 08/03/2012 and the 09/03/2012 and greater night time flows on the 10/03/2012 (Figure 3.9). After the 11/03/2012, differences in day and night time SS concentrations were comparable to those observed before excavation. At this time, the closest excavator was approximately 1.5 km upstream of the site.

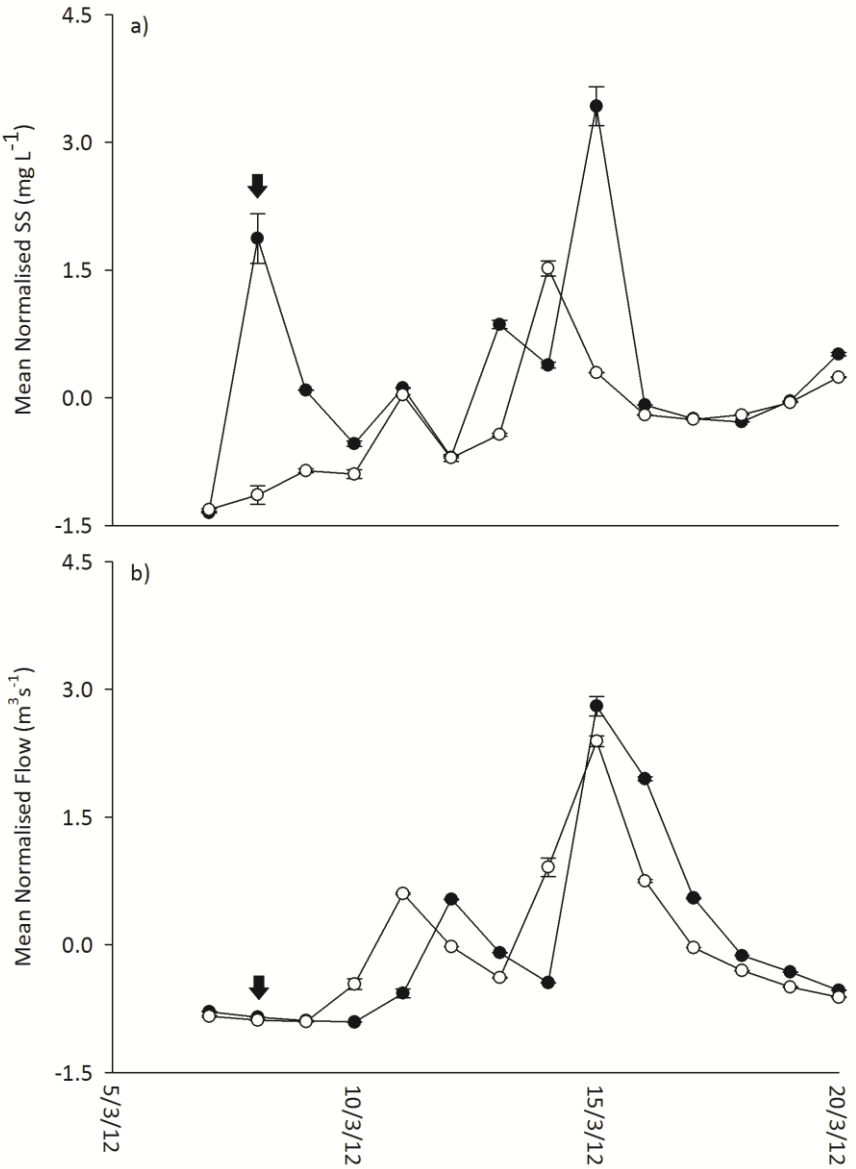

Figure 3.9 a = Daily mean SS concentration (± SE) recorded at Waituna 3 between 08:00 hrs and 20:00 hrs when the excavators were operating (black circles) and between 20:00 hrs to 08:00 hrs when no excavation was performed (white circles). b = Daily mean flow rate (± SE) recorded at Waituna 3 between 08:00 hrs and 20:00 hrs (black circles) and between 20:00 hrs and 08:00 hrs (white circles). Arrows represent the date the excavator moved past the site.

3.4.2.5 Temporal persistence of elevated SS

SS concentrations measured from water samples taken in the four days after excavation (the 14/02/2012 to the 18/02/2012) at Waituna 1 were well above expected levels (calculated from the pre-excavation sediment-flow relationship) (Figure 3.10). SS concentrations during normal flow conditions remained elevated for a significant period after excavation. SS concentrations on the 01/04/2012 (48 days after excavation) exceeded the expected concentration by 10.71 mg L^{-1}. By the 16/04/2012 (63 days after excavation) SS concentrations under normal flow conditions approximated pre-excavation levels. The largest differences in observed versus expected SS concentrations were recorded during high flows. SS concentrations continued to exceed pre-excavation levels during periods of flooding for 77 days after excavation, and SS during a minor flood on the 30/04/2012 exceeded expected concentrations by 228.72 mg L^{-1} (Figure 3.10).

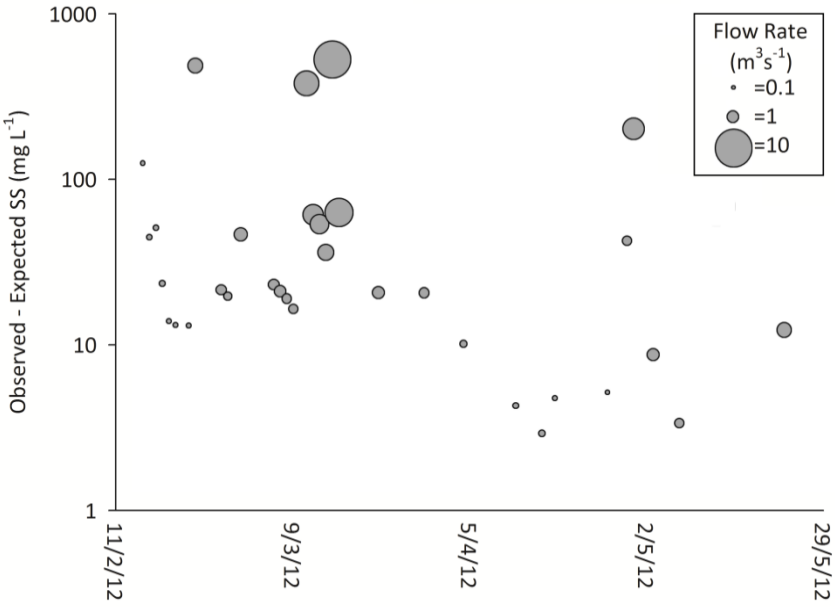

Figure 3.10 Observed - expected SS concentrations at Waituna 1 after excavation. Values represent the deviation of recorded sediment concentrations from the pre-excavation sediment-flow relationship, and were calculated by subtracting the expected SS concentrations (based on flow at time of sample) from the observed values. The area of the circles indicates the flow rate at the time of sample collection (see legend).

3.5 Discussion

3.5.1 Primary findings

The findings of this study findings support predictions that mechanical excavation of macrophytes increases the amount of SS to which fish are exposed, and that elevated levels of SS, particularly during periods of high flow, persist for longer than previously thought. The results of this study suggest that bed disturbance during mechanical excavation of macrophytes is a significant cause of sediment resuspension in New Zealand waterways. Following a large-spatial-scale macrophyte removal operation, statistically significant increases in SS concentrations were observed at six locations in the Waituna catchment. At the most downstream site elevated levels of SS persisted for approximately 77 days after excavation. Point-source sediment pollution from the excavators persisted for only five days in Waituna 1 and Waituna 3, suggesting that elevated SS concentrations after this period were caused by changes in habitat structure and stream morphology.

In contrast to these findings, earlier studies have shown that increases in SS concentration after macrophyte removal in New Zealand streams are short-lived. Wilcock et al. (1998) found that turbidity remained elevated for less than four hours after an 80-m section of a Waikato waterway was excavated. Similar results were recorded by Young et al. (2004), who reported rapid recovery of turbidity following the excavation of an isolated section of a Marlborough drain. Both studies were carried out at small spatial-scales with sites in close proximity to undisturbed sections of waterway. It is possible that the design of these studies led to the underestimation of the effects of macrophyte removal on SS. During large drainage operations downstream reaches may be affected by sediment transport from upstream excavation (Brookes, 1988). However, the impact this has on downstream SS concentrations could not be measured in the studies undertaken by Wilcock et al. (1998) and (Young et al. (2004)).

3.5.2 Small-spatial-scale experimental excavation

3.5.2.1 Drivers of increased SS

The short-term increases in SS concentration observed after small-spatial-scale experimental macrophyte removal in the present study reflect habitat conditions in the study sites before excavation and the macrophyte removal techniques employed by the excavator operator. Increased flow resistance within macrophyte stands reduces near-bed velocity and turbulence, increasing the deposition and retention of fine sediment (Luhar et al., 2008; Jones et al., 2012). Excavation had

not been conducted at these sites in the three years before this study, and large amounts of sediment had accumulated during this period. To extract the root structure of targeted macrophytes effectively, the excavator had to remove the top layer of sediment as well as the vegetation. The perforated bucket used during excavation was designed to let water drain from the extracted vegetation. Consequently, a significant proportion of the excavated sediment was flushed back into the channel and into suspension (pers. obs.).

3.5.2.2 Implications for Fish

While mechanical excavation was found to increase SS concentrations, there is no evidence to suggest that sediment resuspension during and after drainage operations will directly kill fish. Rowe et al. (2009) determined that the median lethal SS concentrations during 24 hours of exposure ranged from 3000 mg L^{-1} to greater than 43,000 mg L^{-1} across the three most sensitive species of New Zealand fish, common smelt (3000 mg L^{-1}), redfin bully (>43,000 mg L^{-1}) and banded kokopu (>43,000 mg L^{-1}). Maximum values recorded in this study exceeded lethal concentrations for common smelt only, and were unlikely to persist for a sufficient time to result in mortality of this species. However, there may be indirect effects on fish physiology associated with increasing SS that were not examined in the present study (Waterman et al., 2011).

In low-flow environments with high organic sediment accretion, biological and chemical oxidation of anoxic organic material can lead to significant reductions in dissolved oxygen during sediment resuspension (Waterman et al., 2011). When determining lethal SS concentrations for native fish, Rowe et al. (2009) controlled for the potential effects of oxygen depletion by 'vigorously' aerating the sediment used in their experiments (Rowe et al., 2009). Consequently, the potential impact of SS on fish mortality may have been underestimated. In situations where anoxic bed sediments are present, oxygen drawdown by organic material may cause mortality at lower concentrations than those suggested by Rowe et al. (2009). Large fish kills in Lake Chilwa, Malawi, have been attributed to the resuspension of sediment with a large organic component, and SS concentrations of 12,860 mg L^{-1} can completely deoxygenate the lake's water column (Bruton, 1985). SS concentrations during small-spatial-scale excavation of Carran 1 were greater than those recorded during fish kills in Lake Chilwa (15,700 mg L^{-1}) with organic matter comprising 43 % of the material in suspension (VSS = 6,800 mg L^{-1}). It is important to note that the effects of sediment suspension on DO may differ between lotic and lentic systems. Although it is unclear how the suspension of this material influenced oxygen conditions in the current study, short-term deoxygenation was observed during mechanical excavation of Waikato streams (see Chapter 5). It is, therefore, reasonable to assume that there was the potential for fish mortality during

mechanical excavation at Carran 1 because of resuspended-sediment-induced hypoxia. This was supported by the large numbers of visibly-stressed giant kokopu recovered during excavation of Carran 1 and a Waikato stream (Chapter 5) (*pers. obs.*).

3.5.3 Large-spatial-scale excavation

3.5.3.1 Drivers of increased SS

In the long-term, the absence of macrophytes may contribute more to increased SS than the initial physical process of removal. The results of this study suggest that long-term increases in SS concentrations at Waituna 1 and Waituna 3 were not directly related to any upstream activities of the excavators. Decreasing macrophyte density increases bed shear stress, thereby reducing the amount of water movement required to mobilise deposited sediment (Jones et al., 2012). Therefore, it is likely that long-term increases in SS concentration observed following excavation were the result of fluvial processes continually re-suspending unconsolidated material previously disturbed during upstream drainage works. This is supported by the large increases in SS concentrations observed during high flow events after excavation (Figure 3.10). The importance of macrophytes as a regulator of sediment resuspension has been demonstrated in lakes (James and Barko, 1994; Madsen et al., 2001; James et al., 2004). In Lake Marsh, Minnesota, significantly more sediment is suspended by wave activity during high winds in years where macrophytes are absent than in years where they are abundant (James and Barko, 1995; Madsen et al., 2001). Using statistical modelling, James et al. (2004) demonstrated that sediment resuspension during high-wind events at another Minnesota lake, Lake Christina, increased with decreasing macrophyte biomass. Although the effects of macrophytes on sediment suspension likley differ between lotic and lentic systems, it is doubtful that SS concentrations return to pre-excavation levels following large-scale drainage operations until all disturbed material is transported out of the system, or sediment compaction and new macrophyte growth increases deposition and retention.

3.5.3.2 Implications for fish

SS concentrations observed following excavation (Figure 3.2) were not sufficient to directly cause fish mortality (Rowe et al., 2009), but may still affect fish abundance and community composition. Juvenile migratory life stages of a number of native fish species have been found to avoid waterways with elevated concentrations of SS (Boubée et al., 1997). The results of this study indicate that increased SS following excavation may limit recruitment of these fishes. SS concentrations regularly exceeded levels required to elicit an avoidance response in juvenile

banded kokopu (25 NTU = 14.9 mg L^{-1} at Waituna 1) , koaro (70 NTU = 61.0 mg L^{-1} at Waituna 1) and inanga (420 NTU = 366.0 mg L^{-1} at Waituna 1) (Boubée et al., 1997) in the 77 days after excavation. Consequently, excavation during juvenile migrations is likely to reduce the abundance of these species. Regular SS concentrations above 120 mg L^{-1} are also thought to reduce the abundance of sensitive species (Rowe et al., 2000), which suggests that SS may be a driver behind reduced fish numbers following excavation of macrophytes (Chapter 2). It is unclear why elevated SS results in avoidance by juvenile native fish and reduces the occurrence of some species, although reduced feeding ability and respiratory performance may be contributing factors (Bruton, 1985; Rowe et al., 2000).

The feeding abilities of some native fishes may be impaired by high levels of SS after macrophyte removal. Even at lower SS concentrations than those recorded in the current study [160 NTU (Quinn et al., 1992; Rowe and Dean, 1998) = 139.4 mg L^{-1} at Waituna 1] the availability of invertebrate prey in New Zealand streams (Quinn et al., 1992), and the feeding ability of juvenile life stages of many native fishes (Rowe and Dean, 1998), is reduced by suspended sediment. Therefore, long-term increases in suspended sediment following excavation are expected to reduce feeding and growth rates of resident fish. This may result in reduced fish abundance as individuals move into more suitable feeding habitat with clearer water (Bruton, 1985; Ryan, 1991; Waters, 1995; Hansen and Closs, 2009). In addition to reduced feeding, elevated SS after excavation may impede respiratory performance in resident fish (Bruton, 1985). To date, the impacts of sediment on the respiration of native fish have not been tested. Severe gill damage caused by chronic exposure to elevated SS concentrations has, however, been observed in the spotfin chub (*Erimonax monachus*) (Sutherland and Meyer, 2007) and coho salmon (*Oncorhynchus kisutch*) (Lake and Hinch, 1999). Physiological stress in high concentrations of suspended sediment has been linked with gill damage in both species (Lake and Hinch, 1999; Sutherland and Meyer, 2007). If sediment has the same impact on the gill structures of New Zealand native species, reduced respiratory performance may contribute to reduced fish abundance after mechanical excavation (Chapter 2).

Deposition of the sediment suspended during and after the macrophyte removal operations in this study may have had a significant impact on downstream fish communities. Deposited fine sediment affects fish growth rates and community structure by interfering with embryonic development (Kemp et al., 2011); reducing dissolved oxygen (Bruton, 1985; Henley et al., 2000); altering habitat suitability and availability (Berkman and Rabeni, 1987; Walling and Amos, 1999; Yamada and Nakamura, 2002; Collins and Walling, 2007); and reducing the availability of invertebrate prey (Wood and Armitage, 1999; Matthaei et al., 2006). Future research should focus

on quantifying the effects of macrophyte removal on downstream sediment deposition and the impact of this on fish.

3.5.4 Future Directions

The findings of this study demonstrate the potential threat posed by current drain management practices to fish communities in New Zealand's lowland waterways. Mechanical excavation is primarily thought to reduce fish abundance through stranding and habitat loss, but our results suggest that elevated SS following macrophyte removal may also alter fish communities. Chapter 4 will focus on quantifying the impacts of SS concentrations observed after excavation on the feeding ability and respiratory performance of fish in New Zealand waterways. Determination of the impacts of sediment resuspension during excavation on the availability of oxygen for resident fish is the subject of Chapter 5.

Chapter 4.0: The effects of suspended sediment on the feeding and respiration of brown trout (*Salmo trutta*)

4.1 Abstract

Sediment resuspension during and after mechanical excavation of macrophytes may have a significant impact on resident fish populations. However, relatively little effort has been made to quantify its influence on the respiratory performance and feeding abilities of fish in New Zealand waterways. In this study the effects of suspended sediment (SS) concentrations [comparable to those recorded after a large-scale macrophyte removal operation (Chapter 3)] on oxygen consumption, (MO_2) and feeding rates of brown trout (*Salmo trutta*) were examined. MO_2 at 0 mg L^{-1}, 150 mg L^{-1}, 300 mg L^{-1}, 450 mg L^{-1} and 600 mg L^{-1} of SS was measured using semi-closed respirometry. Feeding rates at the same SS concentrations were measured using laboratory tank experiments. The results of this study suggest that SS concentrations up to 600 mg L^{-1} have no effect on brown trout MO_2. Conversely, feeding rates were significantly reduced at 450 mg L^{-1} and 600 mg L^{-1}, indicating that sediment concentrations above 450 mg L^{-1} may negatively affect brown trout populations in New Zealand streams.

4.2 Introduction

4.2.1 Mechanical macrophyte excavation and suspended sediment (SS)

Mechanical excavation of macrophytes to maintain drainage outfall in low gradient waterways is common in New Zealand (Hudson and Harding, 2004; Young et al., 2004), and sediment suspended by this activity may have a significant impact on resident fish populations. Flow retardation in dense macrophyte stands increases the deposition of fine sediments, (Luhar et al., 2008; Jones et al., 2012) which are easily resuspended during excavation (Brookes, 1988; Wilcock et al., 1998; Young et al., 2004). The long-term effects of a large macrophyte removal operations on SS concentrations were quantified for the first time in the study presented in Chapter 3. In that study, macrophyte removal in the Waituna Lagoon catchment was followed by significant increases in sediment resuspension, both immediately after excavation [maximum SS during excavation = 15,700 milligrams per litre (mg L^{-1})] and during high flows in the following weeks (maximum SS during high flow = 860 mg L^{-1}). These high levels of sediment were observed because macrophytes are an important regulator of sediment resuspension and prevent erosion of fine particulates (James and Barko, 1994; Madsen et al., 2001; James et al., 2004). Increased bed shear stress following the removal of aquatic plants encourages sediment resuspension by hydrological processes. Consequently, material disturbed during excavation continues to be resuspended until it is flushed out of the system, or new macrophyte growth increases deposition. The time required for this process depends on the morphology, hydrology and sediment properties of the targeted waterway. In the Waituna Creek SS concentrations did not return to pre-excavation levels for approximately 77 days after macrophyte removal (Chapter 3).

4.2.2 Importance of SS to freshwater fishes

Increased SS associated with intensification of agriculture, forestry and mining combined with increasing urbanisation is thought to be a major contributor to the current global decline in freshwater fish biodiversity (Maitland, 1995; Hazelton and Grossman, 2009a). Thus, increased SS concentrations may partially drive reductions in fish abundance after macrophyte removal (Chapter 2). In Chapter 3 the SS levels recorded after mechanical macrophyte excavation did not exceed lethal concentrations identified for brown trout (*Salmo trutta*) (Garric et al., 1990) or native fishes (Rowe et al., 2009). Therefore, it is unlikely that sediment suspended during macrophyte removal in the Waituna catchment was directly responsible for killing resident fish although indirect mortality may have resulted from reduced dissolved oxygen (see Chapter 5). Nevertheless, the abundance and community structure of fish populations in New Zealand streams may be

affected by elevated SS concentrations after macrophyte removal. SS concentrations in the Waituna Creek regularly exceeded levels required to induce avoidance behaviour in juvenile banded kokopu (*Galaxias fasciatus)*, koaro (*Galaxias brevipinnis*) and inanga (*Galaxias maculatus*) (Boubée et al., 1997) in the in the 77 days after macrophyte removal (Chapter 3). Hence, the excavation of macrophytes immediately prior to, or during juvenile migrations may reduce recruitment and abundance of these species in treated reaches (Boubée et al., 1997; Rowe et al., 2000). Avoidance behaviour in response to SS concentrations below those recorded in Chapter 3 has also been observed in salmonids (Scheurer et al., 2009). Consequently, increased SS may result in an exodus of brown trout from recently excavated waterways, and reduce recruitment into these areas. Suspended sediment directly and indirectly affects fish through a variety of mechanisms (Bruton, 1985; Ryan, 1991; Wood and Armitage, 1997), and the exact cause of increased avoidance (Boubée et al., 1997; Scheurer et al., 2009) and reduced abundance (Rowe et al., 2000) in New Zealand streams with high sediment loads is unknown. However, impaired respiratory performance and reduced feeding ability may contribute to the reduction.

Suspended sediment is thought to impede respiratory function in fish by clogging and abrading gill structures, thereby reducing surface area for gaseous exchange (Bruton, 1985; Ryan, 1991; Lake and Hinch, 1999; Sutherland and Meyer, 2007). Although this is a commonly cited effect of SS (Bruton, 1985; Ryan, 1991; Wood and Armitage, 1997; Lake, 2003; Sutherland and Meyer, 2007), to the best of my knowledge severe gill damage caused by sediment has been observed only in spotfin chub (*Erimonax monachus*), white tail shiner (*Cyprinella Galactura*) (Sutherland and Meyer, 2007) and coho salmon (*Oncorhynchus kisutch*) (Lake and Hinch, 1999). Furthermore, the effects of gill damage and associated physiological stress (Lake and Hinch, 1999; Sutherland and Meyer, 2007) on oxygen consumption rates of these species, or any other, has not been tested. Determination of the effects of sediment resuspension on oxygen uptake after excavation is necessary if the drivers of reduced fish abundance after macrophyte removal are to be identified and managed (Chapter 2).

Suspended sediments reduce visibility in the water column by scattering and absorbing light, which can impact on the feeding abilities of fish species that rely on vision to forage (Bruton, 1985; Ryan, 1991; Hazelton and Grossman, 2009a; Hazelton and Grossman, 2009b). Turbidity reduces reactive distance and capture success in many species (Sigler et al., 1984; Sutherland and Meyer, 2007; Hazelton and Grossman, 2009a; Kemp et al., 2011). As a consequence, fish expend more energy when feeding in high-sediment environments (Sigler et al., 1984; Sutherland and Meyer, 2007). Sediment concentrations below those recorded in Chapter 3 have been found to

significantly reduce the feeding ability of the juvenile life stages of many native fishes (Rowe and Dean, 1998), but it is unclear how many adult natives and introduced salmonids are affected by increased turbidity after macrophyte removal. The results reported in Chapter 3 and by Quinn et al. (1992) suggest that macroinvertebrate abundance will be reduced by SS following macrophyte removal. If the feeding abilities of New Zealand fishes are impaired, the depleted prey resources may be insufficient to meet metabolic energy demands. Consequently, fish may leave affected reaches in search of better feeding conditions (Quinn et al., 1992; Wood and Armitage, 1999; Matthaei et al., 2006; Kemp et al., 2011). Determination of the effects of high concentrations of SS on the feeding abilities of fish in New Zealand waterways will facilitate better management of the ecological impacts of macrophyte control.

One fish species found in New Zealand waterways that may be particularly susceptible to the effects of sediment resuspension during and after macrophyte removal is brown trout. Salmonids are known to be sensitive to sediment, and high levels of SS have been shown to increase avoidance behaviour (Bisson and Bilby, 1982; Sigler et al., 1984; Scheurer et al., 2009); reduce feeding success (Sigler et al., 1984); cause gill damage (Sigler et al., 1984; Lake and Hinch, 1999) and increase physiological stress in these fishes (Lake and Hinch, 1999). Although not native to New Zealand waterways, brown trout are highly-valued sport fish, and 59,224 trout fishing licenses were sold across the country (excluding Lake Taupo) in 2011/12 (Unwin, 2013). Therefore, quantifying and publicising the effects of sediment resuspension on brown trout is more likely to generate support for better management of the ecological impacts of macrophyte removal, than similar information on a native species. In addition, the high sensitivity of salmonids to sediment means that setting maximum SS concentrations for the protection of brown trout is likely to provide some level of protection to the less sensitive native species.

4.2.3 Aims

The aim of this study was to examine the effects of sediment resuspension on the respiratory performance and feeding abilities of brown trout. Specifically, oxygen consumption and feeding rates were measured across a range of SS concentrations recorded in the downstream reaches of the Waituna Creek after macrophyte removal (Chapter 3). This study represents the first attempt to quantify the effects of sediment on fish respiratory performance using respirometry, and is the first to measure the effects of SS on the feeding rates of brown trout. Based on the SS concentrations required to cause gill damage in other salmonids [> 40,000 mg L^{-1} in coho salmon (Lake and Hinch, 1999)], it is hypothesised that sediment concentrations lower than 600 mg L^{-1} will have no effect on the oxygen consumption of brown trout. SS concentrations as low as 85 mg

L^{-1} have been found to impair feeding in other salmonid species [steelhead trout (*Salmo gairdneri*) and coho salmon (Sigler et al., 1984)], and it is predicted that SS concentrations between 150 mg L^{-1} and 600 mg L^{-1} will reduce the feeding rates of brown trout.

4.3 Methods

4.3.1 Test fish and husbandry

One-year-old brown trout [mean weight = 3.63 grams (g) ± 0.09; mean length = 73.44 mm ± 0.26] were sourced from the Montrose Fish & Game Hatchery in North Canterbury. Fish were transported from the hatchery to the laboratory in buckets, and upon arrival were transferred to 60-litre (L) flow-through holding tanks (20 fish per tank). Fish were acclimated to laboratory conditions for 10 to 15 days during which water temperature (14˚C) and photoperiod (12 light: 12 dark) were kept constant. Fish were fed with live *Daphnia* spp. every day at 12:00 hours.

4.3.2 Sediment source and concentrations

SS concentrations recorded in the lower reaches of the Waituna Creek after excavation (Chapter 3) were recreated in the laboratory using sediment sourced from the Halswell River (43°39'42.74"S; 172°32'37.47"E). This sediment closely matches sediment found in Waituna Creek in terms of both particle size distribution and organic content (GAL, 2012; Hill-Labs, 2012). Despite these similarities, turbidity in this study was on average 75 % less than in Waituna Creek at the same SS concentrations (Chapter 3). Sediment was oven-dried and sieved through a 250 µm mesh to remove coarse particulates and ensure material remained in suspension during experiments (Boubée et al., 1997). Fish were exposed to the same five sediment treatments [0 milligrams per litre (mg L^{-1}) control; 150 mg L^{-1} (40 NTU); 300 mg L^{-1} (75 NTU); 450 mg L^{-1} (92 NTU); 600 mg L^{-1} (102 NTU)] in the feeding and respiratory components of the study. Treatment levels were selected to span the range of SS concentrations recorded in the lower reaches of Waituna Creek after excavation [maximum SS concentration recorded = 630 mg L^{-1} (Chapter 3)].

4.3.3 Respirometry trials

4.3.3.1 Test apparatus

Oxygen consumption rates (MO_2) of brown trout were measured using a semi-closed respirometry system (Leggatt et al., 2003; Urbina et al., 2012). The system consisted of 10 sealable, cylindrical, glass respirometry chambers [Volume = 196 millilitres (ml)] and five header tanks. Water was circulated between the chambers and header tanks (two respirometry chambers per tank) by pumps (max output = 1200 l h^{-1}). Constant water temperature (14˚C) was maintained by submerging the entire system in a temperature controlled bath regulated by a refrigeration unit (HC-1000A chiller, Hailea, China). Bubblers and air stones kept oxygen saturation at or near 100 % in the header

tanks. Oxygen concentrations (in mg L^{-1}) within the chambers were measured using planar oxygen sensor spots (5mm PSt3 sensor spots, PreSens, Regensburg, Germany) glued to the interior surface and a fibre-optic oxygen transmitter (Fibox 3 LCD, PreSens, Regensburg, Germany). Planar optodes are a relatively recent innovation, but have been used successfully in past respirometry experiments (Warkentin et al., 2007; Cooper et al., 2010; Köster et al., 2010). Data collected by the transmitter were automatically logged on a PC using Presens OxyView Software (V2.04, PreSens, Regensburg, Germany). Two-point calibration of the Presens system was performed daily in oxygen-free water (2 % sodium sulphite solution) and air-saturated water.

4.3.3.2 Experimental procedure

Ten fresh fish were used in each of the six respirometry trials (n_{total} = 60). In each trial two fish were randomly assigned to each sediment treatment ($n_{per\ treatment}$ = 12), and exposed to that concentration of SS for 90 minutes in separate respirometry chambers. Oxygen consumption was measured three times over this period. Prior to sediment exposure fish were left undisturbed for two hours to acclimate to the respirometry chambers (acclimation time determined by preliminary trials). During this time, clean aerated water was pumped through the chambers from the header tanks at a rate of 100 ml min^{-1}. After acclimation, a stock solution of aerated sediment was added to each of the header tanks to bring SS concentrations in the chambers to the desired level. The chambers were then sealed, and oxygen concentrations within were measured. Oxygen concentrations were measured again 20 minutes later, and the difference used to calculate MO_2. The chambers were then opened and refreshed with aerated water at a rate of 100 ml min^{-1} for 10 minutes. This process was repeated between 30 and 50 minutes and between 60 and 80 minutes. To measure and account for sediment oxygen demand and bacterial oxygen consumption in the source water, all trials were repeated without fish in the chambers (controls). At the end of each trial fish were weighed and measured. Respirometry trials were conducted with the permission of the Otago University Animal Ethics Committee, and conducted in accordance with the University of Otago Code of Ethical Conduct.

4.3.3.3 MO_2

For each of the three oxygen measurements taken from individual fish the rate of oxygen consumption [MO_2 (mg O2 g^{-1} min^{-1})] was calculated using:

$$MO_2 = \frac{\left(\frac{\Delta O_2 \times V}{\Delta T}\right) - \Delta BO_2 min^{-1}}{M}$$

Oxygen concentration (O_2) is measured in mg L^{-1}, water volume (V) of the respirometry chamber is calculated as total volume of the chamber minus the volume of the fish [assuming 1 g = 1 ml], time (t) is measured in minutes and mass (M) is measured in grams. The rate of background oxygen consumption (ΔBO_2 min^{-1}) within the chambers did not differ between sampling periods (0-20 min; 30-50 min; 60-80 min) (repeated measures ANOVA $P > 0.05$), and the average of these values were used in the equation above.

4.3.3.4 Data exploration and analysis

The Shapiro-Wilk test was used to test MO_2, weight and length data for the assumption of normality before statistical analyses were conducted. A Levene's test of equality of error variances was also run to test MO_2, weight and length data for the assumption of homogeneity of variance. All assumptions were met. One-way ANOVAs were used to compare the mean weights and lengths of fish between the five different sediment treatments. A two-way repeated measures ANOVA (rep-ANOVA) was used to compare the mean MO_2 of fish exposed to the different sediment treatments. The three MO2 measurements taken per fish were treated as a within-subject factor, and sampling date (block factor) and sediment treatment were treated as between-subjects factors.

4.3.4 Feeding trials

4.3.4.1 Test apparatus

Feeding trials were conducted in five glass tanks in a temperature-controlled room (14˚C). The tanks were 22.5 centimetres (cm) wide by 45.5 cm long, and water depth was 9.75 cm (volume = 10 L). Light was provided by a bank of overhead fluorescent lights. Sediment in the tanks was kept in suspension by overhead stirring units (Eurostar digital, IKA WORKS, Inc. Wilmington, North Carolina) equipped with four-blade propeller stirrers (5 cm diameter) positioned at the centre of the tank, two cm above the floor. The propellers spun at 125 revolutions per minute, the maximum speed at which fish were able to maintain their position in the water column without noticeable increases in exertion. Decreases in SS over the trial periods were negligible [mean decrease in turbidity at; 150 mg L^{-1} = 5.35 % (±0.41), 300 mg L^{-1} = 5.55 % (±0.54), 450 mg L^{-1} = 6.23 (±0.62), 600 mg L^{-1} = 7.29 (±0.59) (measured with a U-50 Series Water Quality Meter, Horiba, Kyoto, Japan)]. Water in the tanks was kept aerated by bubblers and air stones, which also aided sediment resuspension.

4.3.4.2 Experimental procedure

Ten feeding trials were conducted on consecutive days using five fresh fish in each trial (n_{total} = 60). In each trial one fish was randomly assigned to each of the five sediment treatments ($n_{per\ treatment}$ = 10). Preliminary results indicated that 24 hours of starvation was sufficient to eliminate all traces of prey from the gut but fish were not fed for 48 hours before being used in the experiments. After being placed in the tanks fish were left undisturbed in clear water for 22 hours to acclimate to the feeding tanks (Rowe and Dean, 1998). A stock solution of aerated sediment was then added to each tank to bring SS concentrations to the desired level. Fish were left for a further two hours to acclimate to the sediment (Rowe and Dean, 1998). Preliminary trials determined that in clear water trout could eat up to 90 *Daphnia* spp. in 30 minutes. At the start of each experiment 100 *Daphnia* spp. (1-2 mm in length) were placed in each tank. Trout were then left to feed undisturbed for 30 minutes, after which they were quickly netted, and euthanized in 2-phenoxy ethanol. Fish were then weighed and measured. The alimentary canal was dissected out, and *Daphnia* spp. in the buccal cavity, oesophagus, stomach and intestine were counted under a dissecting microscope (Rowe and Dean, 1998). Feeding trials were conducted with the permission of the Otago University Animal Ethics Committee, and conducted in accordance with the University of Otago Code of Ethical Conduct.

4.3.4.3 Data exploration and analysis

The Shapiro-Wilk test was used to ensure feeding, weight and length data met assumption of normality before statistical analyses were conducted. One-way ANOVAs were used to compare the mean weights, lengths and feeding rates (number of prey min^{-1}) of fish exposed to the five different sediment treatments. Fischer's LSD *post hoc* pairwise comparisons were used to evaluate differences between individual treatment groups.

4.4 Results

4.4.1 Respirometry trials

Weights and lengths of fish used in the respirometry experiments did not differ significantly between sediment treatments (Table 4.1) (one-way ANOVA weight, $F_4 = 1.709$, $P = 0.148$, $F_4 = 1.728$, $P = 0.157$). No statistically significant differences in MO_2 were observed between sediment treatments [rep-ANOVA (between-subjects factor), $F_4 = 6.53$, $P = 0.627$, partial eta squared = 0.050] (Figure 4.1). MO_2 differed significantly between the 10 days on which sampling was conducted [rep-ANOVA (between-subjects factor), $F_4 = 3.997$, $P = 0.004$, partial eta squared = 0.286]. Further exploration of the raw data found no correlation between MO_2 and SS concentration ($R = 0.003$). Therefore, it is unlikely that variation in individual MO_2 masked the effects of sediment on the respiratory performance of brown trout.

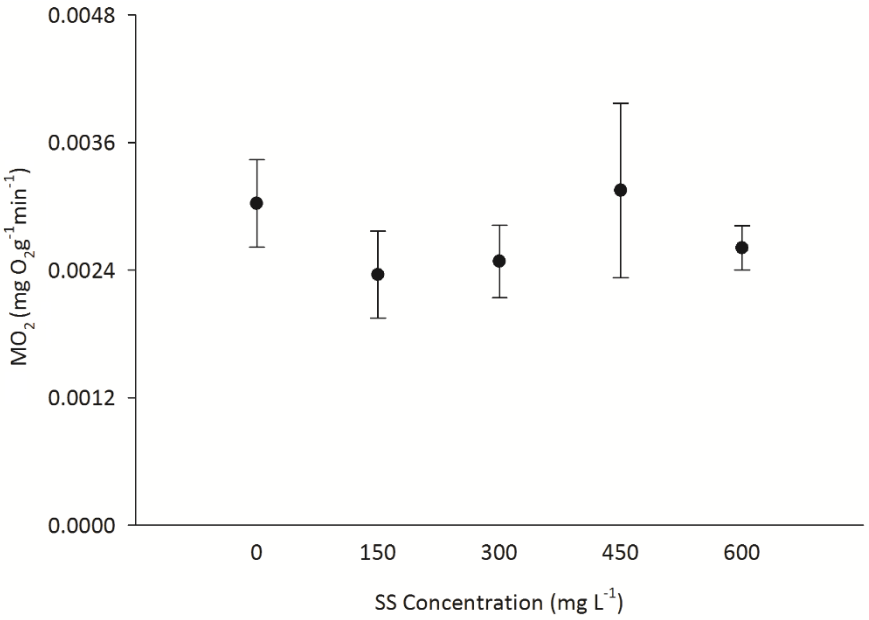

Figure 4.1 Mean (±SE) MO_2 of brown trout exposed to the five SS treatments.

Table 4.1 Mean weights and lengths of fish used in the respirometry trials.

SS conc. ($mg\ L^{-1}$)	Weight (g) Mean ± SE	Length (mm) Mean ± SE
0	3.93 ± 0.29	75.0 ± 0.82
150	3.49 ± 0.26	72.5 ± 0.67
300	4.51 ± 0.30	78.9 ± 0.67
450	3.76 ± 0.31	74.2 ± 0.78
600	3.98 ± 0.24	75.6 ± 0.59

4.4.2 Feeding trials

Table 4.2 Mean weights and lengths of fish used in the feeding trials.

SS conc. ($mg\ L^{-1}$)	Weight (g) Mean ± SE	Length (mm) Mean ± SE
0	2.91 ± 0.16	70.0 ± 1.53
150	3.73 ± 0.99	75.9 ± 2.20
300	3.59 ± 0.10	70.1 ± 4.58
450	2.95 ± 0.62	69.6 ± 1.59
600	3.17 ± 0.81	70.9 ± 1.86

Weights and lengths of fish did not differ significantly between sediment treatments (Table 4.2) (one-way ANOVA weight, $F_4 = 2.084$, $P = 0.099$, one-way ANOVA length $F_4 = 0.998$, $P = 0.418$). Mean feeding rates differed significantly between the different sediment treatments (one-way ANOVA, $F_4 = 6.779$, $P < 0.001$) (Figure 4.2). Feeding rates at 450 mg L^{-1} of SS were significantly lower than at 0 mg L^{-1} and 150 mg L^{-1} (Fisher's LSD pairwise comparison's, 450 mg L^{-1} vs. 0 mg L^{-1} $P = 0.011$; 450 mg L^{-1} vs. 150 mg L^{-1} $P = 0.004$). The mean feeding rate of fish exposed to 600 mg L^{-1} of SS was significantly lower than those exposed to 0 mg L^{-1}, 150 mg L^{-1} and 300 mg L^{-1} (Figure 4.2) (Fisher's LSD pairwise comparison's, 600 mg L^{-1} vs. 0 mg L^{-1} $P < 0.001$; 600 mg L^{-1} vs. 150 mg L^{-1} $P < 0.001$; 600 mg L^{-1} vs. 300 mg L^{-1} $P = 0.003$).

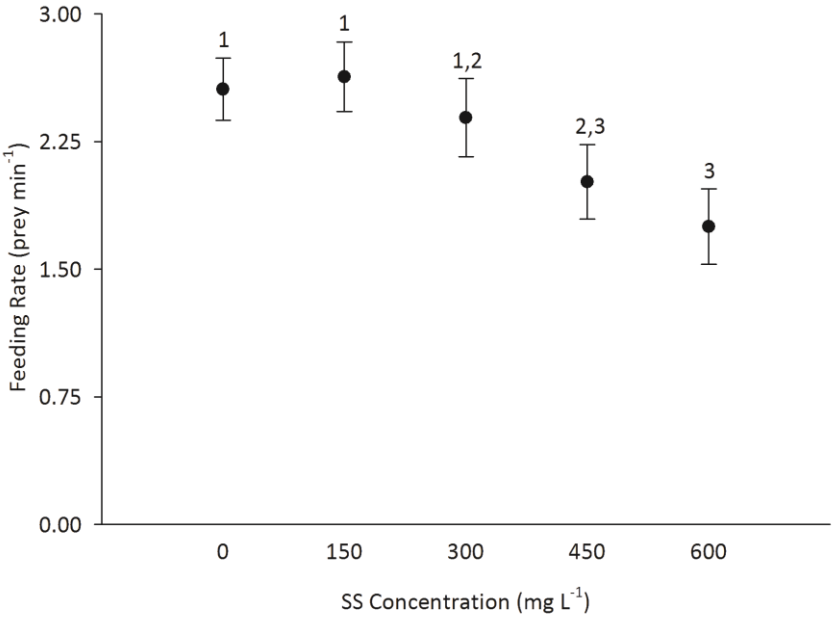

Figure 4.2 Mean feeding rates (±SE) of brown trout exposed to the five SS treatments. Significant differences ($P < 0.05$) between treatments, calculated by Fishers LSD pairwise comparisons, are denoted above (sites that are significantly different do not share a number)

4.5 Discussion

4.5.1 Effects of SS on respiratory performance

It was predicted that SS concentrations up to 600 mg L^{-1} would have no effect on the respiratory performance of brown trout. This prediction was supported by the results, and MO_2 did not differ significantly between fish exposed to the five sediment treatments. The lack of difference in MO_2 across the range of sediment concentrations tested is not surprising in the light of past research on this topic. Gill damage has been observed in salmonids (coho salmon) after short-term exposure to sediment, but only at concentrations exceeding 40,000 mg L^{-1} (Lake and Hinch, 1999). It is likely, therefore, that the sediment concentrations tested in this study were too low and exposure time too short for gill damage and the resulting decreases in MO_2 to occur.

The results of this study may reflect the limitations in its design rather than the effects of SS on brown trout respiration. After real-world drainage operations fish are exposed to elevated SS concentrations for far longer than what they were in this study (1.5 hours). Initially it was intended that brown trout would be exposed to test SS concentrations for 72 hours. However, in order to keep the sediment in suspension for this length of time, a combination of manual agitation, bubblers, pumps and magnetic stirrers was required. This generated a significant source of potential stress, and a number of fish jumped out of the exposure tanks when this was attempted. Hence, the decision was made to forgo an extended exposure period, which may have influenced the results. Gill damage has been observed in minnows (spotfin chub and white tail shiner) after extended exposure to SS concentrations below those tested in the current study (Sutherland and Meyer, 2007). Although the results of Sutherland and Meyer (2007) cannot be applied directly to brown trout, they indicate that longer exposures to SS may have a greater affect on MO_2 than what was observed in this study.

4.5.2 Effects of SS on feeding rate

It was predicted that SS concentrations between 150 mg L^{-1} and 600 mg L^{-1} will reduce the feeding rates of brown trout. This prediction was supported by the results and statistically significant decreases in feeding rates were observed at 450 mg L^{-1} and 600 mg L^{-1} of SS. Concentrations above 450 mg L^{-1} were regularly recorded throughout the Waituna catchment after excavation (Chapter 3), and our results indicate that this activity may affect feeding in resident brown trout populations. Furthermore, the SS concentrations recorded after excavation in Chapter 3 (up to 630 mg L^{-1}) may have had a greater impact on brown trout than the results of

the present study suggest. Reduced feeding performance in high sediment environments is driven by associated increases in turbidity rather than the mass concentration of suspended material (Bruton, 1985; Hazelton and Grossman, 2009a). The relationship between SS and turbidity varies between sediment sources, and is determined by the shape, light absorbing properties and refractive index of suspended particulates (Davies-Colley and Smith, 2001). Turbidities in the sediment solutions used in this study were, on average, 75 % lower than those recorded in the Waituna Creek at the same concentrations (Chapter 3). Therefore, it is likely the sediment concentrations recorded in Chapter 3 had greater influence on feeding rates of brown trout than what was observed in this study.

Suspended sediment most likely reduces the feeding rates of brown trout in two ways (Bilotta and Brazier, 2008). The first is reduced foraging behaviour. In high SS concentrations both steelhead trout and coho salmon cease or reduce surface-feeding behaviours and become less aggressive during feeding attempts (Redding et al., 1987). Similarly, relatively low sediment concentrations (180 mg L^{-1}) have been found to reduce foraging activity of juvenile Atlantic salmon (*Salmo salar*) (Robertson et al., 2007). The second potential cause of reduced feeding in high sediment environments is decreased reactive distance. For visual foragers such as salmonids, turbidity severely impedes feeding ability by reducing the distance at which they can detect prey (Ryan, 1991; Hazelton and Grossman, 2009a; Kemp et al., 2011). Barrett et al. (1992) determined that the reactive distances of rainbow trout (*Oncorhynchus mykiss*) is reduced by 65 % when SS increases turbidity from 5 NTU to 30 NTU. Whatever the cause, reduced feeding in high sediment environments may have significant impacts on brown trout health. Sigler et al. (1984) determined that SS concentrations of 426 mg L^{-1} are sufficient to decrease feeding to the point that steelhead trout and coho salmon growth rates are reduced. Similarly, Shaw and Richardson (2001) found that regular sediment pulses of 704 mg L^{-1} impairs rainbow trout vision, thereby reducing prey capture success, which in turn reduces growth rates. The effects of reduced feeding performance on brown trout health is likely to be amplified by the reduced abundance of invertebrate prey at the SS concentrations recorded after excavation in Chapter 3 (Quinn et al., 1992).

4.5.2.1 Implications of reduced feeding for fish

It is unclear whether reduced feeding rates in high SS concentrations affect brown trout abundance after mechanical excavation of macrophytes. However, a number of salmonid species have been found to leave high sediment environments (Bisson and Bilby, 1982; Sigler et al., 1984; Robertson et al., 2007), allowing them to attain faster growth rates than individuals that remain (Sigler et al., 1984). Therefore, persistent increases in SS concentrations after excavation (Chapter 3) may

reduce brown trout abundance in treated waterways by forcing individuals to leave in search of better feeding habitat and slowing recolonisation.

4.5.2.2 Relevance of results from a management perspective

The results of this study demonstrate the need to reduce sediment resuspension during excavation, but there is still insufficient information for the development of realistic and relevant sediment limits for the protection fish. It is difficult to accurately recreate variable environmental conditions in a laboratory. This presents a major problem when using the results of *ex-situ* studies to determine the effects of pollutants, such as sediment, on fish. The tolerance a fish exhibits to SS is dictated in part by the sediment concentrations to which it has become acclimated (Bisson and Bilby, 1982; Redding et al., 1987). In clear streams resident fish are expected to exhibit a greater sensitivity to sediment than fish of the same species inhabiting naturally turbid waterways (Bisson and Bilby, 1982; Redding et al., 1987; Ryan, 1991; Kemp et al., 2011). In the present study, readily available hatchery trout were used instead of wild fish to provide greater replication and decrease temporal bias. The fish used in this study were first exposed to high concentrations of sediment during the feeding experiments. Thus, despite being the offspring of wild parents, individuals used in this study may have been more sensitive to sediment than their wild counterparts. Further research, focused on quantifying the impacts of SS on trout reared under different natural sediment regimes, is needed if the ecological effects of macrophyte removal are to be understood and managed. Despite this, the findings of the current study are of particular importance as they demonstrate the potential threat to fish health posed by sediment resuspended during macrophyte removal.

Chapter 5.0: Effects of mechanical and chemical macrophyte control on dissolved oxygen in Waikato streams

5.1 Abstract

Dissolved oxygen (DO) is a vital component of water quality that has a significant impact on freshwater fish. Macrophytes are known to be key drivers of DO in eutrophic streams, but relatively little effort has been put into determining how their removal affects oxygen availability. The present study examined the influence of two forms of macrophyte control, mechanical excavation and herbicide application, on diel variations in DO and exposure time of native fish to moderate (< 30 % DO saturation) and severe (< 10 % DO saturation) levels of hypoxia. DO was monitored continuously before and after macrophyte control in 11 streams in the Waikato Region of New Zealand's North Island. Three streams were mechanically excavated, four were treated with herbicide and four were undisturbed (control). Mechanical excavation caused statistically detectable short-term increases in time spent at moderate and severe hypoxia. Changes in diurnal oxygen variations following herbicide application were not consistent, but exposure time to moderate and severe hypoxia increased at some sites. These increases may be associated with increased biological oxygen consumption of the decaying plant material. Results suggest that both forms of macrophyte control increase exposure time to moderate and severe hypoxia, which may reduce fish abundance.

5.2 Introduction

5.2.1 Importance of dissolved oxygen (DO) to freshwater fishes

Dissolved oxygen (DO) is a vital component of water quality, and has a significant impact on aquatic organisms, particularly fish. The amount of oxygen a fish can absorb across fine gill membranes is heavily dependent on environmental oxygen conditions, so reductions in external DO limits the supply of oxygen to body tissues (Dean and Richardson, 1999). Lethal and sub-lethal effects of low DO occur when compensatory mechanisms, such as aerial respiration, surface aquatic respiration (Weber and Kramer, 1983; Kramer, 1987; Stierhoff et al., 2003; McNeil and Closs, 2007), decreased activity and increased gill ventilation (Dean and Richardson, 1999) are no longer effective in maintaining oxygen supply. Chronic exposure to mild hypoxia can retard embryonic development, reduce growth rates and decrease swimming performance (Alabaster and Lloyd, 1982). Hypoxia becomes lethal when oxygen supply is no longer adequate to meet the energy demands essential for life functions (Kramer, 1987).

Sensitivity of fish to DO concentrations is dependent on species and life stage (Alabaster and Lloyd, 1982), and differs markedly between New Zealand fishes. Landman et al. (2005) determined that the median lethal DO concentrations during 48 hours of exposure ranged from 0.54 to 2.65 milligrams per litre (mg L^{-1}) between inanga (*Galaxias maculatus* 2.65 mg/l), common smelt (*Retropinna retropinna;* 1.83 mg L^{-1}) rainbow trout (*Oncorhynchus mykiss;* 1.61 mg $^{L-}$1), common bully (*Gobiomorphus cotidianus;* 0.91 mg L^{-1}) and the shortfin eel (*Anguilla australis*; 0.54 mg L^{-1}). Although Landman et al. (2005) and Dean and Richardson (1999) disagree about the relative sensitivities of different species, both authors suggested that extended exposure to DO saturation below 30 %, and acute exposure to concentrations below 10 % saturation will result in significant fish mortality in New Zealand streams [values extrapolated from temperatures and DO concentrations (mg L^{-1}) presented by Dean and Richardson (1999) and Landman et al. (2005)]. Macrophytes are a potential driver of DO in eutrophic streams, yet relatively little effort has been put into determining how their removal affects oxygen availability for fish.

5.2.2 Macrophytes and DO

The influence of macrophytes on oxygen conditions is mostly dictated by biomass. Even in non-eutrophic systems, photosynthesis and respiration by plants typically drives a diel cycle in DO (Walling and Webb, 1992). Oxygen concentrations increase during photosynthetic activity by day and decrease with respiration at night (Walling and Webb, 1992; Wilding et al., 2012). As plant density increases, diurnal oxygen variation becomes more pronounced, and in streams where nutrient enrichment and habitat modification has allowed macrophytes to proliferate, it can be extreme (Wilcock et al., 1999b; Wilcock and Nagels, 2001). The flow-retarding properties of macrophytes in weed-clogged streams further amplify their influence on DO concentrations. As macrophyte density increases, reductions in flow and increased water depth (Hearne and Armitage, 1993) lowers the rate of oxygen exchange between the atmosphere and the surface of the water (Thyssen and Erlandsen, 1987; Wilcock et al., 1999a). Without reaeration to mitigate against increased plant respiration, DO depletion in New Zealand streams can reach potentially lethal levels in macrophyte clogged streams (Dean and Richardson, 1999; Wilcock et al., 1999b; Landman et al., 2005). As forty-three percent of the 425,000 kilometres of waterway in New Zealand drain catchments that have been modified for agriculture (Van Bunnik et al., 2007), de-oxygenation driven by macrophytes may pose a significant threat to native fish.

5.2.3 Macrophyte removal and DO

Although regular removal of macrophytes should, in theory, improve oxygen conditions in high productivity streams, the limited research does not support this. In Switzerland it has been found that oxygen conditions in eutrophic streams improved negligibly after weed cutting despite increased reaeration (Kaenel et al., 2000). Similarly, stream metabolism was unaffected by experimental weed cutting in the Waikato region of New Zealand, and diurnal oxygen variations were unchanged despite improved atmospheric oxygen transfer (Wilcock et al., 1999a). The small spatial scale of both these studies, however, meant that it was not possible to determine whether these results were influenced by upstream oxygen conditions (Wilcock et al., 1999a; Kaenel et al., 2000). It is also possible that the removal of plant material simply increased algal productivity (Sand-Jensen, 1983; Biggs and Close, 1989; Uehlinger et al., 1996; Uehlinger and Naegeli, 1998; Kaenel et al., 2000), leaving biological oxygen supply and demand unchanged (Kaenel et al., 2000). It is currently unclear how macrophyte removal techniques that are more disruptive than weed cutting influence oxygen conditions in lowland New Zealand streams.

5.2.4 Potential for mechanical excavation to reduce DO

Bed disturbance during mechanical excavation of macrophytes may result in significant short-term oxygen consumption by oxidisation of organic material in suspended sediments. The increased flow resistance within macrophyte stands reduces near-bed water velocity and turbulence, increasing the deposition and retention of fine particulates (Luhar et al., 2008; Jones et al., 2012). Benthic accumulation of these particulates limits oxygen transfer between the water and stream bed to the top two to five millimetres (mm) of deposited sediment, promoting anoxic conditions in the layers below (Simpson et al., 1998). Below this layer, anaerobic microbial decomposition mineralises and reduces particulate organic matter to soluble intermediates. When the sediment is resuspended into an oxic environment these intermediates are rapidly consumed through bacterial metabolism or chemical oxidization, which reduces DO in the water column (DiToro, 2001; Waterman et al., 2011; Krevs and Kucinskiene, 2012). Bed disturbance during excavation often leads to the resuspension of large amounts of this anoxic sediment previously deposited in the weed bed (Brookes, 1988; Wilcock et al., 1998). In Chapter 3 suspended sediment concentration increased by 120,000 % (13 to 15,600 mg L^{-1}) after experimental excavation in a New Zealand stream, a large proportion of which was organic matter (43 %). Given the impact of this form of macrophyte control on sediment resuspension, it could be expected that the excavation of plant material would be accompanied by significant draw-downs in DO, but this is yet to be rigorously tested.

5.2.5 Potential for herbicide application to reduce DO

The use of herbicides to control macrophytes may also lead to significant draw-downs in DO because of macrophyte decay. Not only is photosynthetic oxygen production diminished following macrophyte dieback (Brooker and Edwards, 1973; Newbold, 1975), but decaying plant material can also be a significant DO sink (Brooker and Edwards, 1973; Newbold, 1975). Biological oxygen demand is increased following macrophyte dieback due to aerobic microbial decomposition of plant tissues (Jewell, 1971; Godshalk and Wetzel, 1978). When macrophyte biomass has developed to the extent that reaeration cannot prevent oxygen depletion during decomposition, the water column can become completely deoxygenated within a few days of herbicide application (Jewell, 1971). Although a number of studies focused on quantifying the impact of herbicide application on DO conditions in reservoirs and small lakes during the 1970s (Jewell, 1971; Brooker and Edwards, 1973; Newbold, 1975), to my knowledge there have been no comparable studies carried out in streams.

5.2.6 Aims

The aim of this study was to examine the impact of macrophyte control on dissolved oxygen conditions in lowland streams. Specifically, I aimed to determine how mechanical and hebicidal removal of macrophytes influenced diurnal variations in DO and the occurrence of moderate [< 30 % DO saturation (significant fish mortality expected after 48 hours of exposure)] and severe hypoxia [< 10 % DO saturation (significant fish mortality expected after 1 hour of exposure)] (Dean and Richardson, 1999; Landman et al., 2005). This study is the first in-depth comparison of the effects of the two forms of macrophyte control on DO, in order to elucidate their relative impacts on fish health. Based on findings presented in Chapter 3 and Waterman et al. (2011), it is hypothesised that rapid deoxygenation will occur during excavation, corresponding with the resuspension of anoxic bed sediments. It is also predicted that herbicide application will result in decreased daily DO minima and maxima, as decomposing plants photosynthesise less and respire more. Periods of hypoxia are also predicted to occur after herbicide application (Jewell, 1971).

5.3 Methods

5.3.1 Study area

This study was carried out in 11 low gradient streams throughout the Waikato Region of New Zealand's North Island (**Figure 5.1**). The region's largest lotic system is the Waikato River, which drains Lake Taupo and most of the Waikato Plains as it makes its 425 kilometre (km) journey from Taupo to Port Waikato. The dominant land use in the area is dairy production, which has had a significant impact on the waterways in the region. A large number of streams have been extensively modified during the conversion of wetlands to pasture (Gibbs, 2006). In addition, agricultural production has degraded water quality to the extent that 85 % of Waikato River tributaries, with the exception of the Waipa and Taupo tributaries, have unsatisfactory levels of nitrogen (WRC, 2008). This nutrient enrichment has allowed macrophytes such as reed sweet grass (*Glyceria maxima*), parrot's feather (*Myriophyllum aquaticum*), oxygen weed (primarily *Egeria densa*) and water pepper (*Persicaria* spp.) to proliferate. To provide drainage to surrounding farmland, the Waikato Regional Council (WRC) conducts regular macrophyte control in over 1800 km of waterway at a cost of $1.5 million per annum (Hudson and Harding, 2004). Despite poor water quality (WRC, 2008) and regular anthropogenic disturbance (Hudson and Harding, 2004; Gibbs, 2006), waterways in the Waikato region are important to native fish. Threatened species present include giant, shortjaw and banded kokopu (*Galaxias argenteus*, *G. postvectis* and *G. fasciatus* respectively), koaro (*G. brevipinnis*), inanga, black mudfish *(Neochanna diversus*), bluegill and redfin bully (*Gobiomorphus hubbsi*, *G. huttoni*), lamprey (*Geotria australis*) and longfin eels (*A. dieffenbachii*) (Allibone et al., 2010).

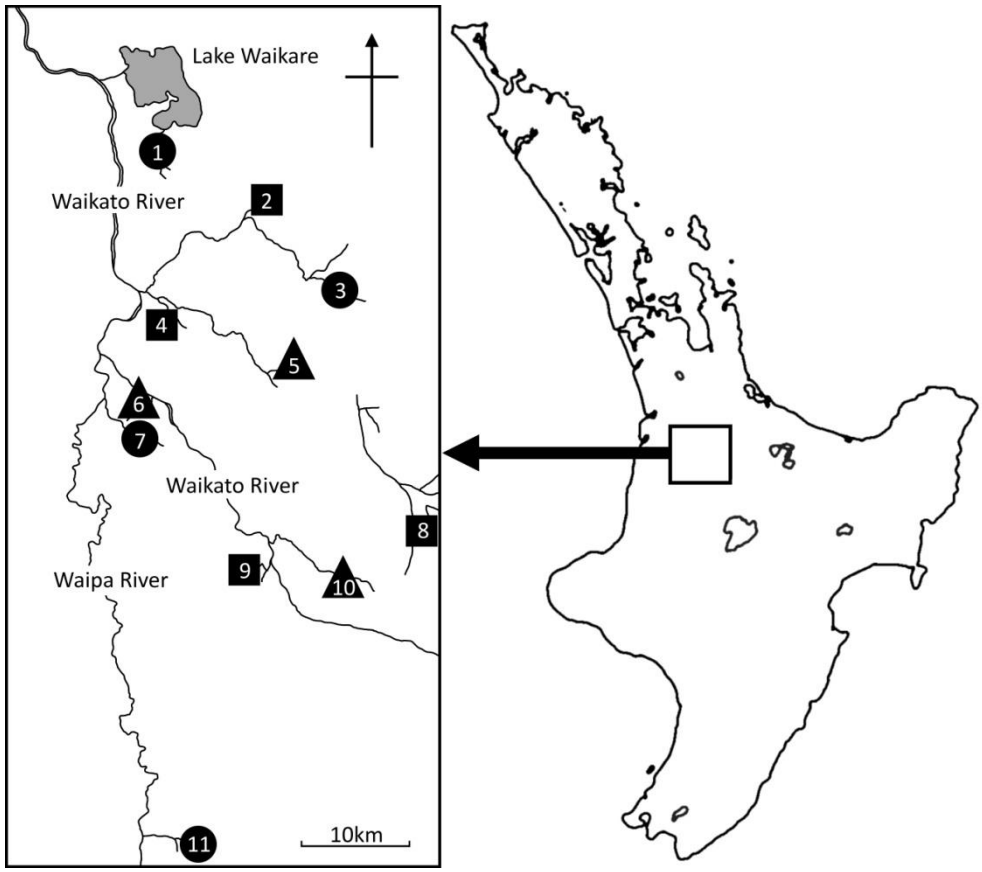

Figure 5.1 Study sites. Triangles and circles represent mechanically and chemically treated sites while squares represent control sites. Numbers represent 1: Frost Road. 2: Pukemokemoke, 3: Tauhei, 4: Kerie Road, 5: Russell Outlet, 6: Affco, 7: Horotiu, 8: Berry, 9: Lake, 10: Hautapu, 11: Mangawhero.

5.3.2 Study sites

Eleven sampling streams were selected in the region based on drain maintenance operations scheduled by the WRC. Between the 01/02/2013 and the 01/04/2013, three streams (hereafter referred to as Affco, Russell Outlet, Hautapu) were mechanically excavated by the WRC, and herbicide was applied in four others (hereafter referred to as Tauhei, Horotiu, Mangawhero and Frost Road) (Figure 5.1). The control group of streams (hereafter referred to as Kerie Road, Pukemokemoke, Lake and Berry) (Figure 5.1) were selected from untreated streams in the region based on similarities in stream morphology and habitat to the treatment streams. Drought-related limitations in site availability meant that stringent site selection criteria could not be imposed, and all the streams excavated or treated with herbicide over the study period were sampled. As a result, there were some differences in site characteristics. Mean macrophyte coverage ranged from 0 % to 100 %, exceeding 40 % in all but three of the study streams (Affco, Kerie Road and Russell Outlet). Average stream widths ranged from 1.1 metres (m) to 6.0 m, and mean water depth ranged from 13.9 centimetres (cm) to 64.7 cm. All streams drained dairy pastures, and had comparable catchments with similar topography and land use.

Data collection in each of the sampling streams was limited to 150-m study sites. In streams where macrophyte control was conducted, sites were located at the downstream end of the treated section of waterway. This ensured that the effects of upstream drainage works were also recorded, and that the entire site would be treated in one day. The locations of the control sites were selected to allow easy access without interrupting nearby dairy operations.

5.3.3 Site surveys

To determine the impact of the different macrophyte control techniques on habitat quality and stream morphology, study sites were surveyed before and after (mechanical = three weeks after; herbicide and control = five weeks after) macrophyte control (control sites surveyed on the first and last day DO was measured). Five transects placed at 30 m intervals along each site were used to estimate: total macrophyte coverage, submerged macrophyte coverage, emergent macrophyte coverage and the relative abundance of the key macrophyte species. Coverage was measured by visually estimating the percentage of stream bed obscured by the canopy or shoots of macrophytes in a 1-metre strip downstream of each transect. At each transect, stream width was measured at water surface level, and average depth was calculated from measurements taken at 30 %, 50 % and 70 % of transect width.

5.3.4 Mechanical excavation and herbicide application

The macrophyte control operations were commissioned by the WRC, and the methods could not be standardised to suit the requirements of study design. Subsequently, the length of waterway treated varied markedly between streams, as did the timing of the operations and the methods employed.

5.3.4.1 Mechanical excavation

Excavators used in this study were equipped with one of two attachments. Russell Outlet and Hautapu were excavated using a non-perforated bucket that removed both plant material and silt from the streambed (primary goal of excavation at Russell Outlet was silt removal) (Figure 5.2a). Affco was excavated using a weed rake (Figure 5.2b). This attachment removed plant material but extracted relatively little silt.

Figure 5.2 Photograph of the two attachments used while excavating the study sites. a) Depicts a standard bucket similar to that used to excavate silt and plant material from Russell Outlet and Hautapu. b) Depicts a weed rake similar to that used to excavated plant material from Affco.

5.3.4.2 Herbicide application

Two herbicides were used in the study streams, glyphosate and endothall (applied as the liquid formulation Aquathol® K). Glyphosate kills macrophytes by inhibiting enzymatic production of certain amino acids required for protein synthesis. Endothall is a protein phosphatase inhibitor that disrupts mitosis leading to cell death (Tresch et al., 2011). Glyphosate was used to control emergent species [predominantly water pepper and reed sweetgrass (*Glyceria maxima*)] at Horotiu, Tauhei and Frost Road, while endothall was used to control oxygen weed (*Egeria densa*) at Mangawhero. Herbicide was dispensed at all sites using a pressurised spray gun. Glyphosate was applied directly onto vegetation from the bank at a rate of 10 ml L^{-1}while endothall was sprayed on the water surface at a rate of 1 ml L^{-1} to be delivered to submerged macrophytes through dispersal.

5.3.5 Spatial variability of dissolved oxygen

Changes in spatial oxygen variability were determined through spot measurements made before and after macrophyte control. The number of site visits was limited by the geographical distance between the sites and the low number of personnel involved in the study. Consequently, before and after data were available only for six sites (Mechanical: Affco, Russell Outlet; Chemical: Tauhei, Horotiu; Control: Berry, Pukemokemoke). During each visit five transects were placed at irregular intervals along the site. DO concentration and temperature were recorded at 30 %, 50 % and 70 % of the channel width at each transect using a Hach HQ40d portable multi-parameter meter and an IntelliCAL™ LDO101 Luminescent/Optical Dissolved Oxygen Probe. All measurements were taken from the vertical centre (50 % depth) of the water column to ensure results were not influenced by sediment oxygen consumption or by atmospheric oxygen transfer at the surface. A record was made of the habitat in which each measurement was taken (open water, macrophyte stand or an algal mat), and a note was made if the surrounding vegetation was dead or dying. To minimise bias resulting from temporal fluctuations in DO, all spot measurements were taken between 10:00 hours and 15:00 hours, with the exception of before data collected from Russell Outlet, which was collected at 08:00 hours.

5.3.6 Continual dissolved oxygen monitoring

Zebra-tech D-Opto loggers were used to measure DO and temperature every ten minutes for at least five days before and 18 days after macrophyte control, with one exception. The decision was made by WRC to clear Russell Outlet just three days before excavation began, therefore data were

recorded for three days before excavation instead of five. There were enough loggers to monitor DO in two control sites at all times (with the exception of the period from 13/03/2013 to 18/03/2013, when all but one logger was needed in the treatment sites) ensuring that natural changes in DO conditions following chemical and mechanical macrophyte control could be isolated from treatment effects. At each site continual DO data were used to calculate daily minimum and maximum saturations and the daily percentage of DO measurements registering < 30 % saturation (moderate hypoxia) and < 10 % saturation (severe hypoxia).

Before deployment, a solution of sodium sulphite (0 % DO saturation) and a completely aerated water sample (100 % DO saturation) were used to calibrate the DO loggers against the Hach HQ40d portable multi-parameter meter used to take spot measurements (± 2.5 %). Optical measurements of DO are accurate to 1% between 0.01 % and 100 % saturation (manufacturers specifications). Accuracy drops to ± 5 % at 250 % saturation (manufacturers specifications), above which, readings are no longer reliable (Wilcock et al., 2011). Values above 250% saturation were regularly recorded in sites with high algal biomass (Affco, Mangawhero and Lake) and were not included in analyses. Although the manufacturer states that sensor drift is less than 1 % per year, sensors were retrieved at least once every two weeks, and checked against 0 % and 100 % DO solutions to ensure accuracy (Wilcock et al., 2011). In all cases sensor drift was negligible (measurements made at 0 % and 100 % saturation within ± 2.5 % of Hach handheld meter), and recalibration was only required when the loggers were relocated.

5.3.7 Analysis

5.3.7.1 Data exploration and transformations

Shapiro-Wilk tests were used test data for the assumption of normality before statistical analyses were conducted. No transformations were required to meet the assumption of normality for DO or site characteristic data. All statistical analyses were carried out using SPSS Statistical Software version 21.0.0 (International Business Machines Corporation, Armonk, NY, USA).

5.3.7.2 Statistical tests

5.3.7.2.1 Site characteristics

Two-way repeated-measures ANOVA (hereafter referred to as rep-ANOVAs) were used to determine the effects of the different macrophyte control treatments on mean depth, width, and macrophyte coverage. Due to the low number of sites sampled in this study (mechanical n=3;

chemical n=4; control n=4) each of the five transects sampled at each site were treated as replicates. The two measurements taken at each transect (before and after treatment) were treated as the within-subject factor (time), and treatment (mechanical, chemical, control) treated as the between-subjects factor.

5.3.7.2.2 Macrophyte control and time spent at moderate and severe hypoxia

For each site, data on time spent at moderate (percentage of DO measurements below 30 % saturation) and severe hypoxia (percentage of DO measurements below 10 % saturation) were calculated for seven three-day sampling periods to use in a before-after-control-impact comparison (BACI) (Stewart-Oaten et al., 1986). In mechanically and chemically treated sites the first sampling period consisted of the three days prior to treatment, and the remaining six periods encompassed the 18 days after treatment. In the control sites, data were split into seven three-day periods beginning the first full day of data collection. The number of sampling periods reflects the maximum number of days on which data were available across all sites. Sampling periods were limited to three days to reflect the maximum number of days before macrophyte control on which data were available for all sites. To control for differences in DO regime between study sites, time spent at moderate and severe hypoxia was normalised across the sampling periods (normalisation adjusts the variance among sampling periods to give a common scale across all sites). Normalisation of DO data was calculated separately for each study site using the mean and standard deviation of the seven sampling periods [(x - mean)/ standard deviation].

Rep-ANOVAs were used to test whether the occurrence and persistence of moderate and severe hypoxia was affected by the different macrophyte control treatments. Two separate rep-ANOVAs were run. Normalised time at moderate hypoxia (percentage of DO measurements less than 30 % saturation) was the repeated measure in the first rep-ANOVA, and normalised time at severe hypoxia (percentage of DO measurements less than 10 % saturation) was the repeated measure in the second. In both analyses the sampling period (time) was treated as the within-subject factor, individual study sites were treated as replicates, and treatment (mechanical, chemical, control) was treated as the between-subjects factor. Low statistical power meant biologically relevant effects [partial eta-squared > 0.3 (Cohen, 1988)] could not be detected by the rep-ANOVAs when the alpha (significance) level was set at 0.05. Consequently, time and time*treatment interaction effects were considered statistically significant when $P \leq 0.10$. In both rep-ANOVAs, the effects of the different macrophyte control treatments over time were examined by planned simple within-subjects contrasts comparing time spent in hypoxia in the first sampling period (before macrophyte

control) with time spent in hypoxia in each of the six remaining sampling periods (after macrophyte control).

5.3.7.2.3 Macrophyte control and spatial variability in DO

Spot DO measurements taken before and after macrophyte control were used to determine the relative impacts of the different forms of macrophyte control on spatial variability of DO during the day. Sample variance in the 15 DO spot measurements taken from each site before and after macrophyte control was used as a measure of spatial variability and was calculated using the following equation:

$$Variance = \frac{\sum(x_i - \mu)^2}{n - 1}$$

Here *x* represents the 15 individual measurements of DO saturation taken at the site, μ represents the mean of all measurements taken at the site and *n* represents the number of DO measurements.

A two-way rep-ANOVA was used to determine the effects of the different macrophyte control treatments on spatial DO variability. Sample variance was the repeated measure; time (before and after macrophyte control) was treated as the within-subject factor; individual study sites were treated as replicates; and treatment (mechanical, chemical, control) was treated as the between-subjects factor.

5.3.7.3 Descriptive analyses of changes in daily DO patterns

Plots of DO changes over time were used to further examine the effects of mechanical and chemical macrophyte control on DO between sites exposed to different treatments. For each site, daily percentages of DO measurements below 30 % and 10 % saturation were plotted, and used to outline the potential impacts of the different forms of macrophyte control on resident fish. Changes in diurnal DO patterns were also analysed descriptively by plotting daily minimum and maximum DO saturations from sites subjected to the different treatments (Perna and Burrows, 2005).

5.4 Results

5.4.1 Site characteristics

Mean depth differed significantly between the different treatment groups [rep-ANOVA (between-subjects factor), $F_2 = 23.45$, $P < 0.001$, partial eta-squared = 0.541]. Across all transects mean depth was significantly reduced after macrophyte control [rep-ANOVA (within-subjects factor) time, $F_1 = 4.916$, $P = 0.02$, partial eta-squared = 0.129]. The significant time*treatment interaction [rep-ANOVA (within-subjects factor) time*treatment, $F_2 = 5.639$, $P = 0.007$, partial eta-squared = 0.220] indicates that changes in mean depth before and after macrophyte control were dependant on macrophyte control treatment. As Figure 5.3 shows, this significant interaction effect was driven by differences between the chemical and mechanical treatments, in which mean depth decreased, and the control treatment in which mean depth increased.

Mean width differed significantly between the different treatment groups [rep-ANOVA (between-subjects factor), $F_2 = 5.843$, $P = 0.006$, partial eta-squared = 0.226]. Across all transects mean width did not differ significantly before and after macrophyte control [rep-ANOVA (within-subjects factor) time, $F_1 = 0.040$, $P = 0.843$, partial eta-squared = 0.001]. In addition, no significant time*treatment interaction was found [rep-ANOVA (within-subjects factor) time*treatment, $F_2 = 0.826$, $P = 0.445$, partial eta-squared = 0.040], indicating that changes in mean width before and after macrophyte control were not dependant on macrophyte control treatment. This is supported by the relatively small changes in mean width observed in all three treatment groups after macrophyte control (Figure 5.3).

Mean macrophyte coverage differed significantly between the different treatment groups [rep-ANOVA (between-subjects factor), $F_2 = 7.719$, $P = 0.001$, partial eta-squared = 0.278]. Across all transects mean macrophyte coverage was significantly reduced after macrophyte control [rep-ANOVA (within-subjects factor) time, $F_1 = 31.985$, $P < 0.001$, partial eta-squared = 0.444]. However, no significant time*treatment interaction was found [rep-ANOVA (within-subjects factor) time*treatment, $F_2 = 2.478$, $P = 0.097$, partial eta-squared = 0.110], indicating that changes in mean macrophyte coverage before and after macrophyte coverage were not dependant on macrophyte control treatment. This is supported by the similar decreases in mean macrophyte coverage observed in all three treatment groups (Figure 5.3).

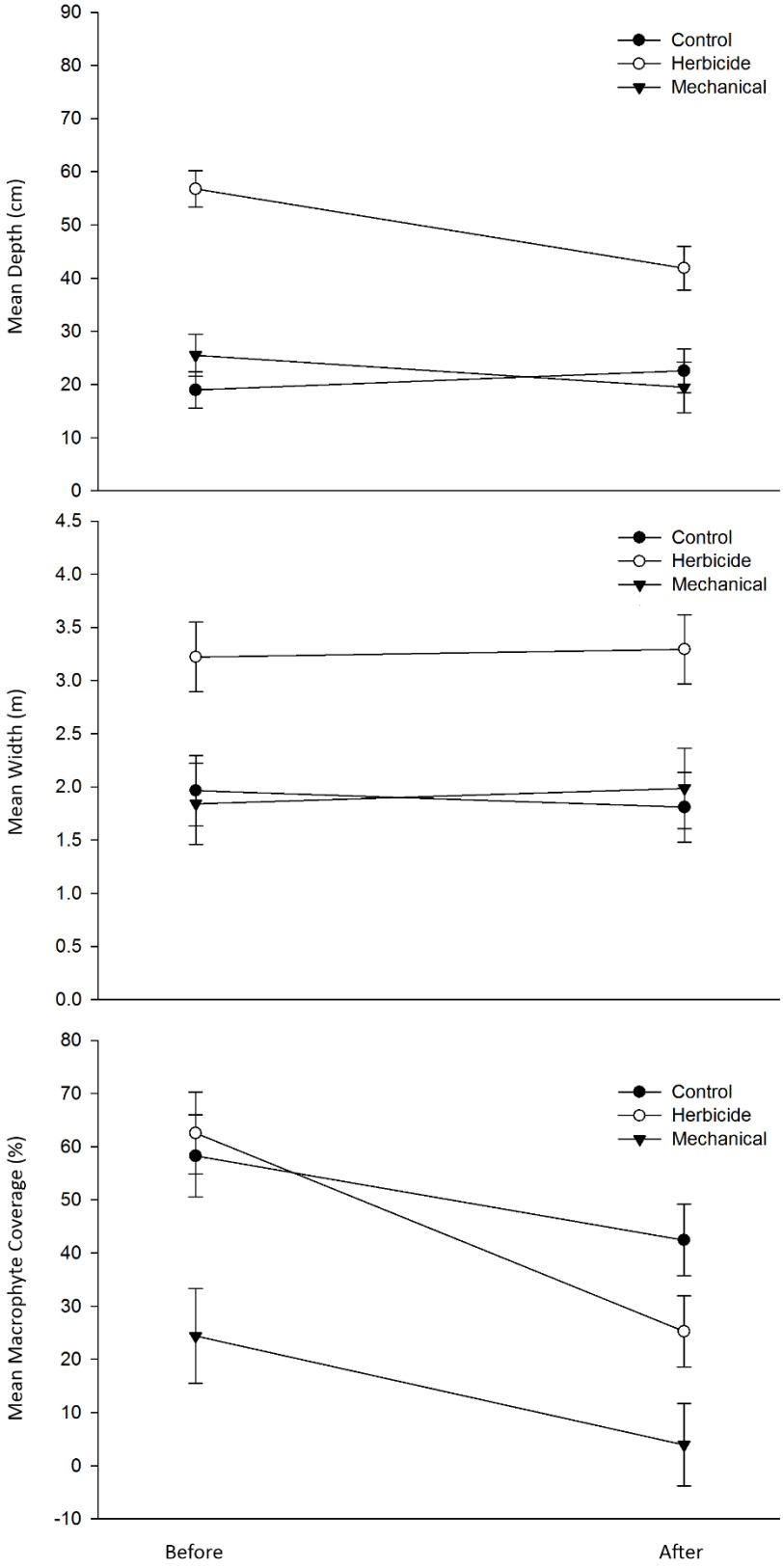

Figure 5.3 Mean (±SE) depth (a), width (b) and macrophyte coverage (c) before and after macrophyte control in the control (black circles), chemical (white circles) and mechanical treatment groups (black triangles).

5.4.2 The influence of mechanical excavation and herbicide application on DO

5.4.2.1 Moderate and severe hypoxia

Time spent in moderate hypoxia (the percentage of DO measurements below 30 percent saturation) differed significantly between treatment groups [rep-ANOVA (between-subjects factor), F_2 = 3.418, P = 0.085, partial eta-squared = 0.461]. Repeated measures ANOVA showed that over the entire study period, both the main effect of time (sampling period) and effect of the time*treatment interaction was not significant [rep-ANOVA (within-subjects factor) time, F_6 = 1.894, P = 0.101, partial eta-squared = 0.191; time*treatment, F_{12} = 1.082, P = 0.396, partial eta-squared = 0.213]. However, planned simple within-subjects contrasts showed that mean time spent at moderate hypoxia was significantly greater 1 - 3 days after treatment than 1 - 3 days before treatment (planned simple within-subjects contrasts, time, F_1 = 25.821, P = 0.001, partial eta-squared = 0.763). Furthermore changes in time spent at moderate hypoxia between these two sampling periods differed significantly between the treatment groups (planned simple within-subjects contrasts, time*treatment, F_2 = 3.243, P = 0.093, partial eta-squared = 0.448). Figure 5.4 shows that the differences between these sampling periods were driven by a marked increase in time spent at moderate hypoxia in mechanically excavated sites 1 - 3 days after treatment.

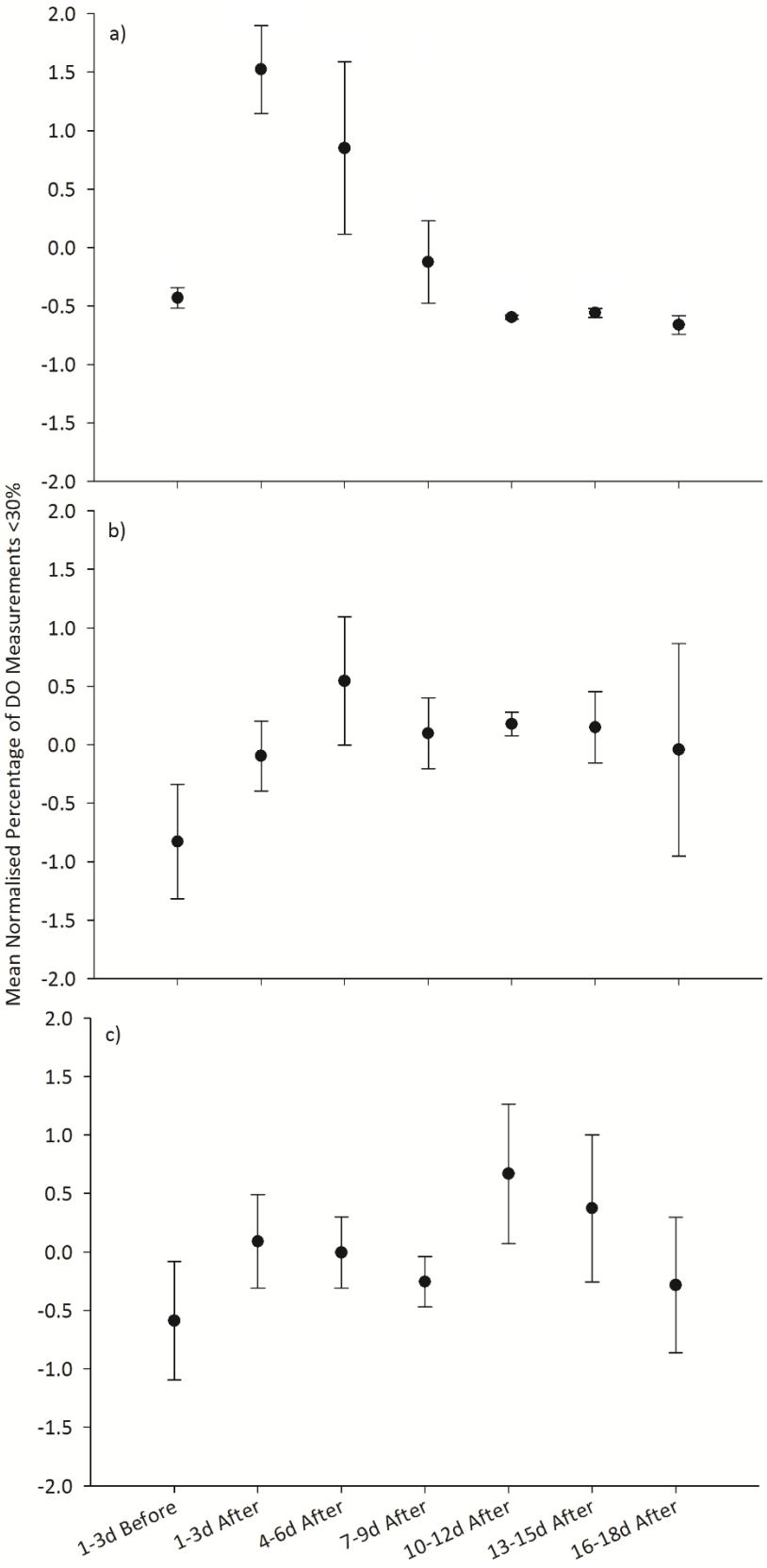

Figure 5.4 Mean normalised percentage of DO measurements below 30 percent saturation (± SE) recorded in seven three-day periods beginning three days prior to treatment (a = mechanical; b = chemical; c = control).

Time spent in severe hypoxia (the percentage of DO measurements below 10 percent saturation) differed significantly between the different treatment groups [rep-ANOVA (between-subjects factor), $F_2 = 3.418$, $P = 0.085$, partial eta-squared = 0.461]. Repeated measures ANOVA showed that although the main effect of time (sampling period) was significant the overall effect of the time*treatment (macrophyte control treatment) interaction was not [rep-ANOVA (within-subjects factor) time, $F_6 = 1.083$, $P = 0.099$, partial eta-squared = 0.193; time*treatment, $F_{12} = 1.434$, $P = 0.184$, partial eta-squared = 0.264]. However, planned simple within-subjects contrasts showed that mean time spent at severe hypoxia was significantly greater 1 - 3 days after treatment than 1 - 3 days before treatment, and that the changes in time spent at moderate hypoxia between these two sampling periods differed significantly between the treatment groups (planned simple within-subjects contrasts, time, $F_1 = 22.262$, $P = 0.002$, partial eta-squared = 0.736; time*treatment, $F_2 = 10.418$, $P = 0.006$, partial eta-squared = 0.723). Figure 5.5 shows that the differences between these sampling periods were driven by a marked increase in time spent at severe hypoxia in mechanically excavated sites 1 - 3 days after treatment.

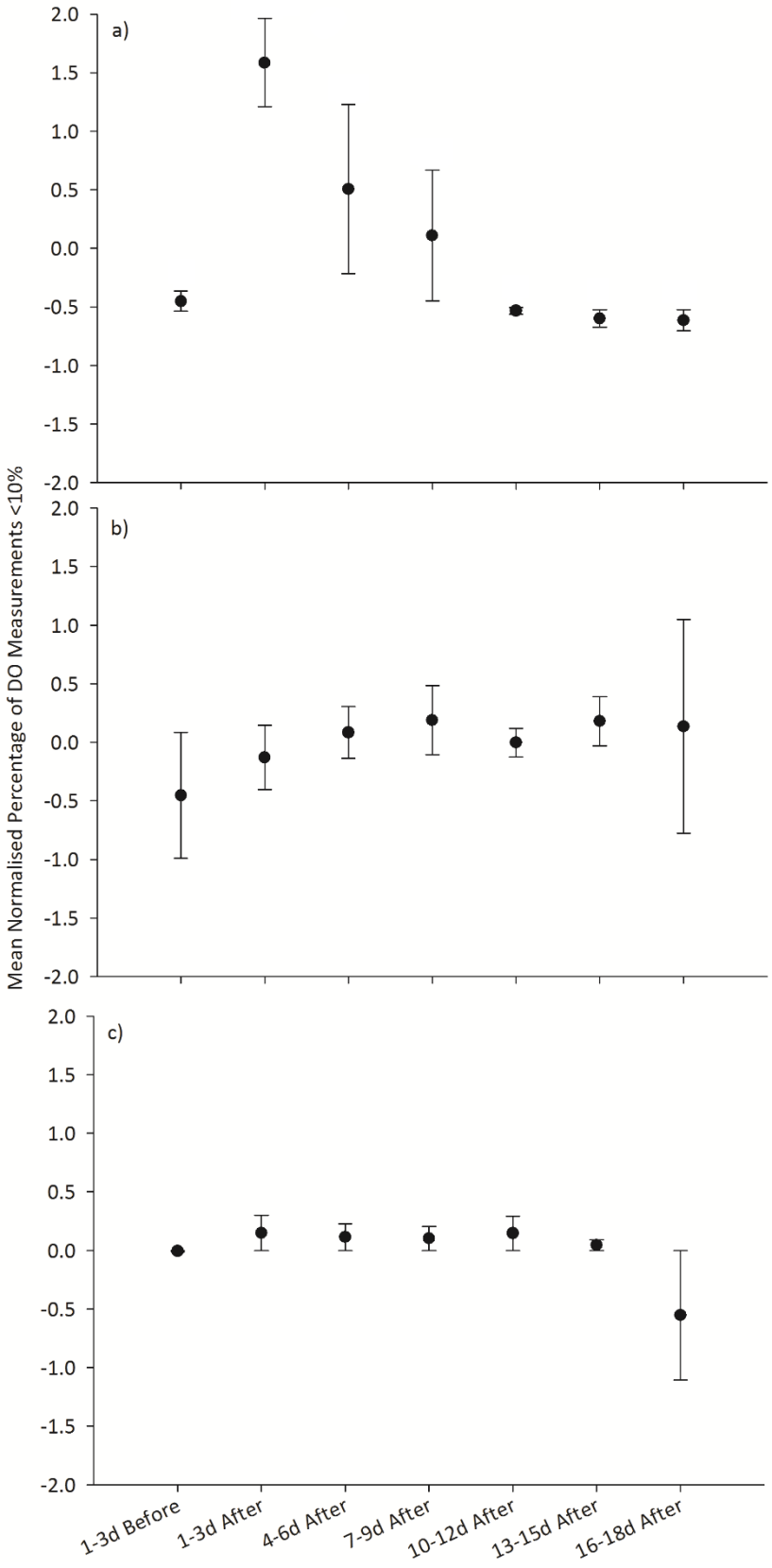

Figure 5.5 Mean normalised percentage of DO measurements below 10 percent saturation (± SE) recorded in seven three-day periods beginning three days prior to treatment (a = mechanical; b = chemical; c = control).

Following excavation the daily percentage of DO measurements below 30 percent saturation (moderate hypoxia) (Figure 5.6) and the daily percentage of DO measurements below 10 percent saturation (severe hypoxia) (Figure 5.7) increased markedly at all mechanically excavated sites. The magnitude and persistence of these increases varied between sites (Figure 5.6 and Figure 5.7). Increased time at moderate and severe hypoxia at excavated the sites was not reflected in values collected from the control sites at the same time (Figure 5.6 and Figure 5.7).

After herbicide application the percentage of DO measurements below 30 percent saturation (moderate hypoxia) increased markedly in three of the four chemically treated sites (Tauhei, Horotiu and Mangawhero) (Figure 5.6). Following herbicide application, the daily percentage of DO measurements below 10 percent saturation (severe hypoxia) increased in two sites (Tauhei and Horotiu) (Figure 5.7). The magnitude and persistence of increases in DO measurements below 30 percent and 10 percent saturation varied between sites. Increases in the daily percentage of DO measurements below 30 percent saturation similar to those recorded in two of the sites (Tauhei and Horotiu) were also observed in a control site (Pukemokemoke) (Figure 5.6). Increases in the daily percentage of DO measurements below 10 percent saturation in the chemically sites were not reflected in values collected from the control sites at the same time (Figure 5.7).

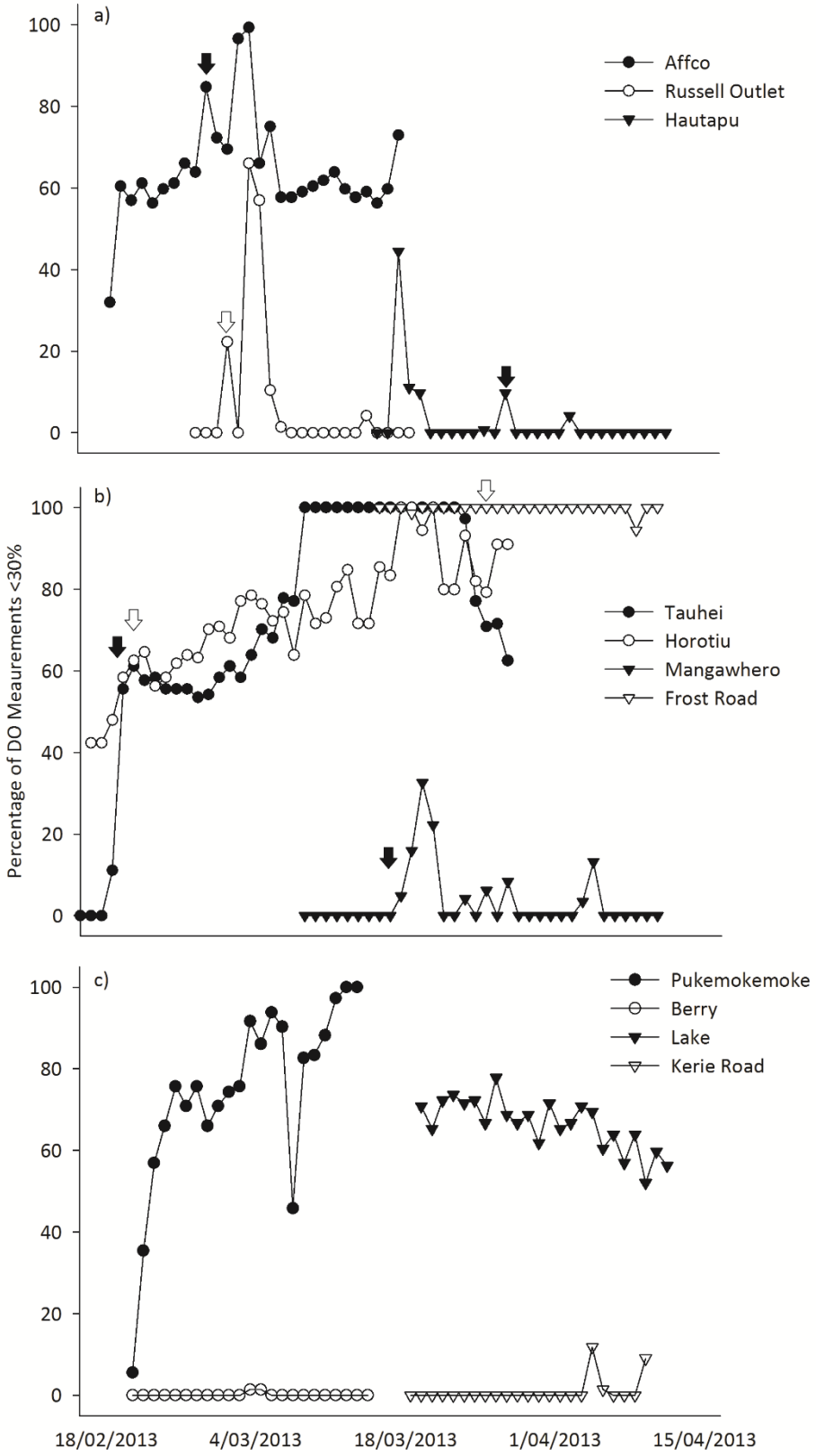

Figure 5.6 Percentage of DO measurements below 30 % saturation plotted by day (a = mechanical; b = chemical; c = control). Date of treatment is illustrated by arrows.

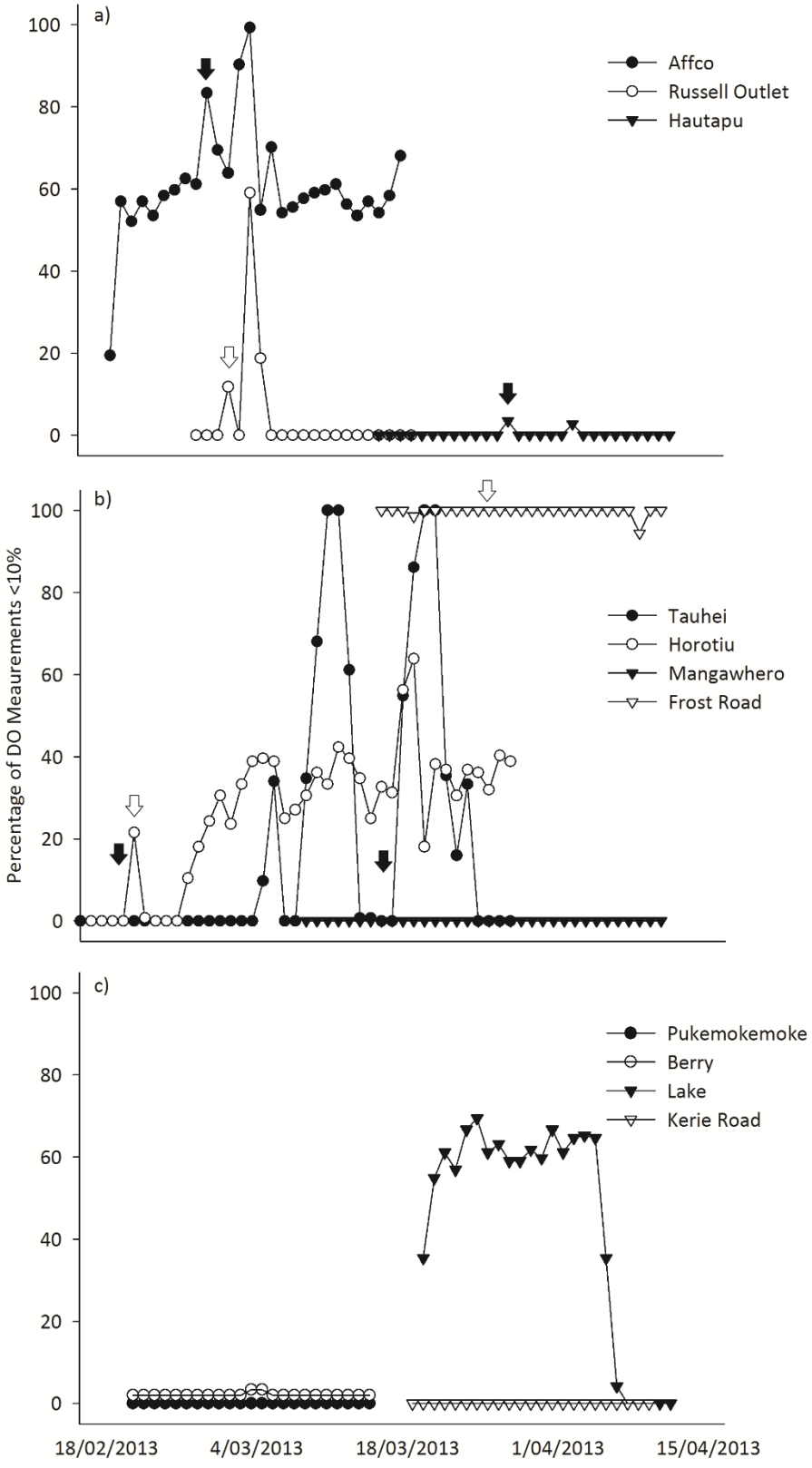

Figure 5.7 Percentage of DO measurements below 10 % saturation plotted by day (a = mechanical; b = chemical; c = control). Date of treatment is illustrated by arrows.

5.4.2.2 Daily DO minima and maxima

After mechanical excavation minimum daily DO saturation decreased in two of the three sites (Russell Outlet and Hautapu) and remained unchanged in the remaining site (Affco) (Figure 5.8). For the most part decreases in daily DO minima at the excavated sites were not reflected in values collected from the control sites at the same time (Figure 5.8). After mechanical excavation, maximum daily DO saturation decreased in two of the three sites (Affco and Russell Outlet) and remained unchanged in the remaining site (Hautapu) (Figure 5.9). Similar decreases in daily DO maxima were not observed over this period in the control group (Figure 5.9).

After herbicide application minimum daily DO saturation decreased at three of the four sites (Tauhei, Horotiu and Mangawhero) and remained unchanged in the remaining site (Frost Road) (Figure 5.8). For the most part, decreases in daily DO minima after herbicide application were not reflected in values collected from the control sites at the same time (Figure 5.8). Daily maximum DO saturations remained unchanged in three of the four sites after herbicide application (Horotiu, Mangawhero and Frost Road) and initially increased in the remaining site before gradually decreasing below pre-treatment levels (Figure 5.9). Similar decreases in daily DO maxima were not observed over this period in the control group (Figure 5.9).

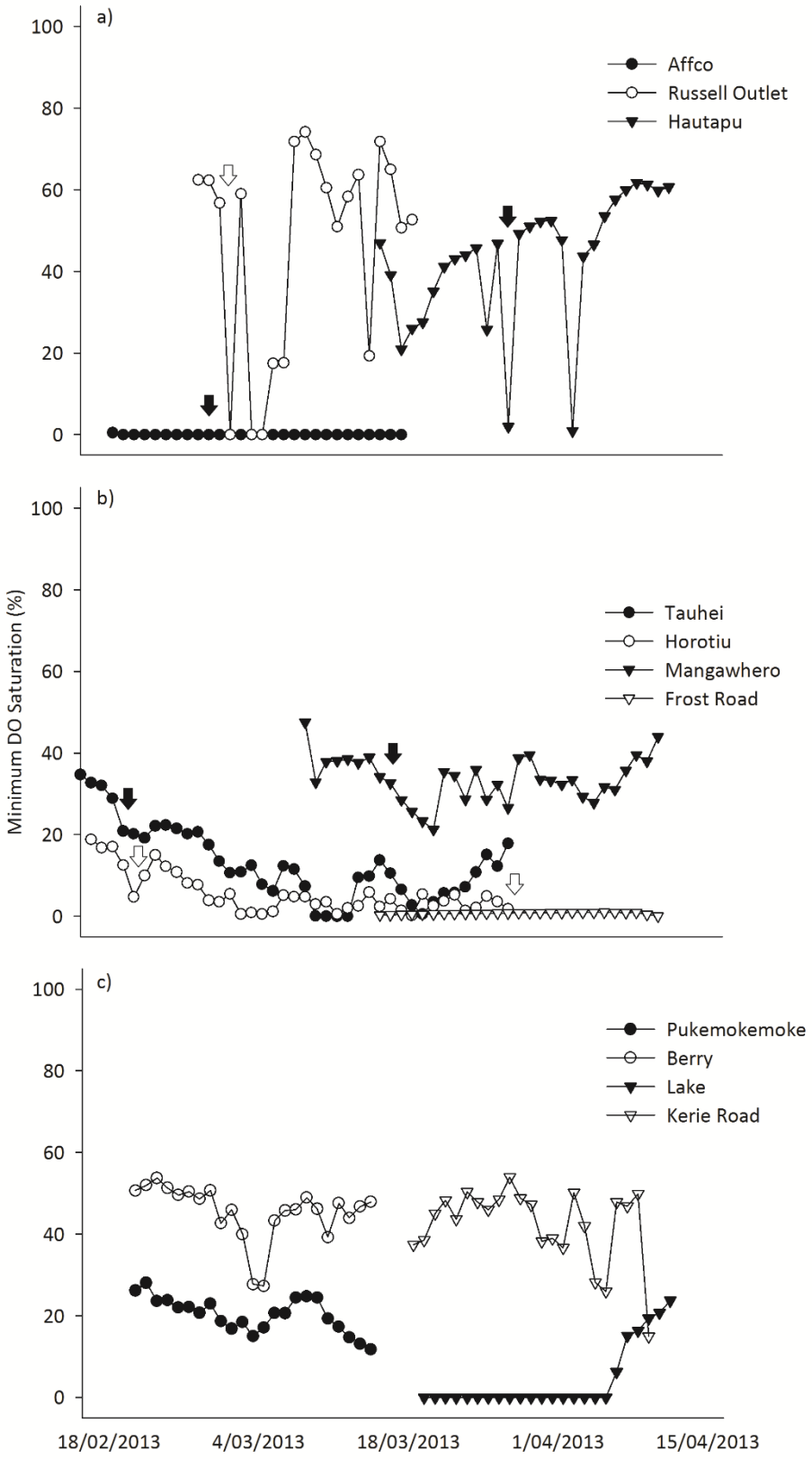

Figure 5.8 Minimum DO saturation plotted by day (a = mechanical; b = chemical; c = control). Date of treatment is illustrated by arrows.

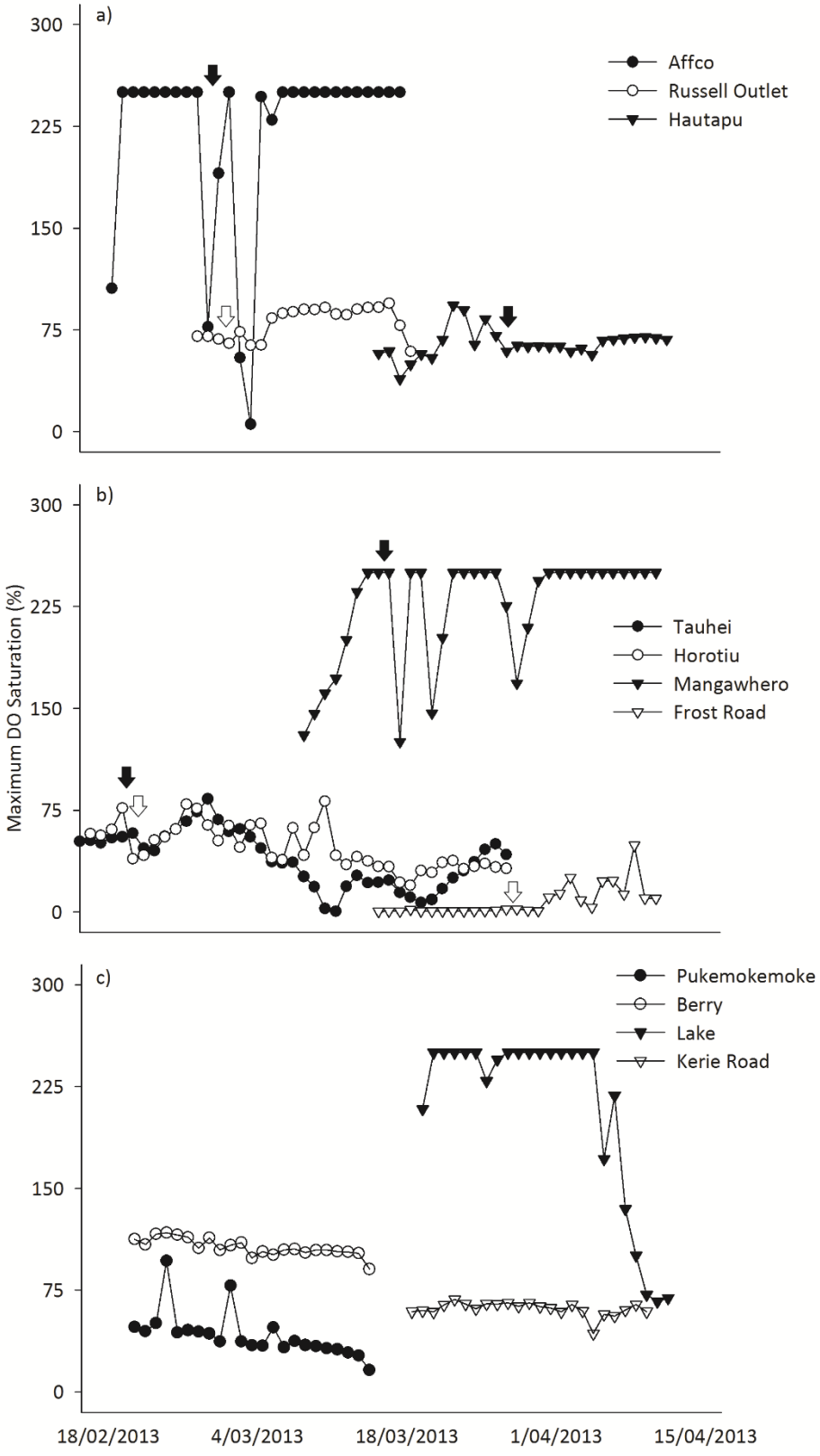

Figure 5.9 Maximum DO saturation plotted by day (a = mechanical; b = chemical; c = control). Date of treatment is illustrated by arrows.

5.4.2.3 Macrophyte control and spatial variability in DO

Overall, macrophyte control treatment type was found to have a medium size effect [partial eta-squared = 0.1 - 0.3 (Cohen, 1988)] on mean DO sample variance [partial eta-squared = 0.397]. That this effect was not detected as statistically significant [rep-ANOVA (between-subjects factor), $F_2 = 0.987$, $P < 0.468$], is most likely because of the low replication in the study design (mechanical n=2; chemical n=2; control n=2). Across all sites time was found to have a medium size effect on mean DO sample variance [partial eta-squared = 0.247]. However, this effect was not detected as statistically significant [rep-ANOVA (within-subjects factor) time, $F_1 = 0.985$, $P = 0.394$]. The time*treatment interaction was also found to have a medium size effect on mean DO sample variance (partial eta-squared = 0.402). Although not statistically significant [rep-ANOVA (within-subjects factor) time*treatment, $F_2 = 1.009$, $P = 0.462$], the size of this effect indicates that changes in the sample variance of DO measurements taken before and after macrophyte control were dependant on macrophyte control treatment (Cohen, 1988). As Figure 5.10 shows, the medium size interaction effect was driven by differences between the mechanical treatment, where DO sample variance decreased markedly after macrophyte control, and the chemical and control treatments, where mean DO sample variance increased slightly.

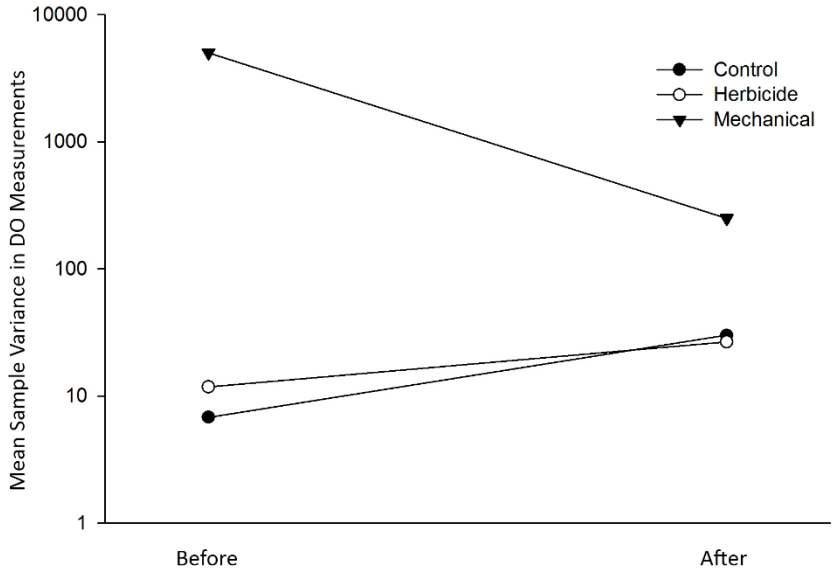

Figure 5.10 Mean sample variance in DO measurements taken before and after macrophyte control in the control (black circles), chemical (white circles) and mechanical treatment groups (black triangles).

5.5 Discussion

5.5.1 Macrophyte control and DO

5.5.1.1 Mechanical excavation

The results of this study suggest that the mechanical excavation of macrophytes depletes DO in the short term. This supports the hypothesis that excavation would significantly increase the time that fish are exposed to hypoxia. Although statistically detectable differences in time spent at moderate (< 30 percent DO saturation) and severe hypoxia (< 10 percent DO saturation) were limited to a three-day period immediately after excavation, altered diurnal oxygen cycles persisted for as long as eight days in some sites. Furthermore, spatial DO variability decreased following mechanical excavation. This indicates that the availability of refugia, within which fish could escape hypoxia during this period, may have been reduced by mechanical excavation. Rapid mortality (≤ 45 minutes) of hypoxia sensitive species such as inanga and smelt in low DO has been observed in laboratory studies (Dean and Richardson, 1999; Landman et al., 2005). Fish housed in a stable laboratory setting may exhibit different levels of DO sensitivity to their wild counterparts. Nevertheless, the findings of this study highlight the potential for significant mortality following excavation in the wild. This is supported by the large numbers of visibly-stressed (surface breathing, loss of equilibrium etc.) giant kokopu seen recovered during excavation of Russell Outlet (pers. obs.).

In contrast to these findings, earlier studies have shown that the removal of macrophytes has a relatively small impact on DO. Wilcock et al. (1999a) found that stream metabolism, and daily DO minima and maxima did not change in Waikato streams following weed cutting. Similar results were found by Kaenel et al. (2000), who determined that weed raking had little impact on DO conditions in eutrophic streams in Switzerland. Both studies were relatively small scale with sites in close proximity to undisturbed sections of waterway. It is possible that the design of these studies did not allow them to detect differences in DO before and after macrophyte removal due to upstream oxygen production and consumption. Alternatively, inconsistencies between the results of the current study and those reported in Wilcock et al. (1999a) and Kaenel et al. (2000) may reflect the differences in the techniques used to remove macrophytes. In the studies reported by Wilcock et al. (1999a) and Kaenel et al. (2000) plant material was cut either manually or mechanically then removed from the waterway, leaving the stream bed undisturbed (Kaenel et al., 2000). On the other hand mechanical excavation has a significant effect on the stream bed

(Brookes, 1988; Wilcock et al., 1998). It is possible that it is this bed disturbance that drove the observed changes in DO after excavation.

In this study changes in macrophyte coverage did not differ significantly between the mechanical, chemical and control treatments, which suggests factors other than reduced photosynthetic oxygen production drove increased hypoxia in mechanically excavated sites. Consequently, it is likely that decreases in DO were caused by an unmeasured factor associated with bed disturbance. Mechanical excavation results in the resuspension of large amounts of sediment (Brookes, 1988; Wilcock et al., 1998). Once suspended, bacterial metabolism and chemical oxidization of the anoxic component of this sediment rapidly consumes oxygen, reducing DO in the water column (DiToro, 2001; Waterman et al., 2011; Krevs and Kucinskiene, 2012). Although it is not possible to conclusively identify the drivers behind DO patterns observed in the present study, significant organic sediment resuspension has been recorded during excavation of other New Zealand streams (Chapter 3). It is, therefore, reasonable to assume that sediment resuspension may have contributed to increased hypoxia after excavation.

5.5.1.2 Herbicide application

It was predicted that herbicide application would reduce daily DO minima and maxima as respiration increased and photosynthesis decreased during plant decomposition. This prediction was not supported by these results, and the influence of herbicide application on DO differed between sites. As predicted, daily DO minima decreased in three of the four study sites following macrophyte removal. Conversely, with the exception of one site, herbicide application had no effect on daily DO maxima. As predicted, increases in moderate and severe hypoxia were observed in a number of sites after herbicide application. However, unlike mechanical excavation, changes in time spent at moderate and severe hypoxia after herbicide application differed between sites and were not detected in the statistical analyses conducted. Despite this, the occurrence of persistent severe hypoxia after herbicide application indicates that this form of macrophyte control has the potential to increase fish mortality.

Changes in diel oxygen cycles following herbicide application are common in reservoirs and small lakes (Jewell, 1971; Brooker and Edwards, 1973; Newbold, 1975). Hypoxic events, like those recorded at two of the sites included in this study, have previously been observed following herbicide application, and are most likely to occur in eutrophic systems with high macrophyte biomass and low reaeration rates (Jewell, 1971; Almazan and Boyd, 1978). Changes in diel DO cycles, and time spent in hypoxia were most likely caused by a combination of reduced

photosynthetic oxygen production and increased biological oxygen demand, due to aerobic microbial decomposition of plant tissue (Jewell, 1971; Newbold, 1975; Almazan and Boyd, 1978; Godshalk and Wetzel, 1978).

Limitations of the study design may explain the differences in DO patterns observed between sites following herbicide application. Stream morphology, habitat and water quality differed between streams, as did the herbicide used, so sites were not true replicates of the same treatment. Glyphosate was used at Tauhei, Horotiu and Frost Road. Morphology and habitat characteristics were similar across these sites, and oxygen patterns at Horotiu and Tauhei were approximately the same following herbicide application. DO saturation at Frost Road was 0 % before treatment, which meant there was no potential for oxygen drawdown during plant decomposition. Mangawhero was a much larger waterway than the other three streams with greater reaeration to counteract DO consumption by decomposing macrophytes. In addition, it is known that endothall is ineffective at killing the *Egeria densa* that comprised the majority of the macrophyte community in this site (Hofstra and Clayton, 2001), which is supported by the relatively small changes observed in DO at the site. Drought in the Waikato in early 2013 meant the majority of planned drainage works were postponed. The four chemically treated streams included this study were the only waterways in which the WRC applied herbicide during study period. The inclusion of Mangawhero (endothall) and Frost Road (0 % DO) in the study probably resulted in an underestimate of the impacts of herbicide application on diel DO cycles, an underestimate of the extent of hypoxia and failure to detect uniform changes over time. It could be argued that these sites should have been excluded from the study, but with no replacements available, there was no benefit in doing so, particularly as their inclusion provided some insight into the effects of herbicide application on DO in different systems.

5.5.2 Primary findings and recommendations

These findings are of particular importance as they demonstrate the potential threat posed by current drain management practices to fish communities in New Zealand's lowland waterways. Mechanical excavation is primarily thought to reduce fish abundance through stranding, habitat loss and reduced food availability. The results of this study suggest that hypoxia immediately after excavation may also be a contributing factor. The recovery of stranded fish from the banks of excavated waterways is considered best practice in streams containing high-value species. The findings of the present study suggest that the scope of recovery operations should be expanded to include the relocation of fish exhibiting obvious signs of hypoxic stress (surface breathing, loss of

equilibrium etc.) within the waterway. Despite the limitations imposed on the study design, the results suggest that herbicide application also has the potential to increase hypoxia to the extent that unless native fish move out of treated waterways, mortality will increase. Unfortunately low water levels in the Waikato in 2013 meant it was not possible to conduct net-based fish surveys before and after herbicide application in this study. Future research focused on monitoring fish populations following herbicide application is needed to measure the ecological impacts of chemical macrophyte control accurately, and to identify streams that are at greatest risk.

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Chapter 6.0: General discussion

6.1 Disturbance

Disturbance regimes are important organisers of community structure in riverine ecosystems, exerting a strong influence over species composition and abundance (Sousa, 1984; Reice et al., 1990). Intensification of land use has had a significant impact on disturbance regimes in freshwater ecosystems, making them some of the most threatened in the world (Maitland, 1995; Dudgeon et al., 2006). However, the roles of many anthropogenic activities in the disturbance of freshwater ecosystems are not understood. Macrophyte removal is a common practice (Hudson and Harding, 2004), but the study described in this thesis is one of the first to thoroughly examine its impacts on freshwater fish communities. For disturbance to have occurred, "ecosystem, community, or population structure" must be disrupted and "resources, availability of substratum, or the physical environment", must be changed (White and Pickett, 1985).

This research found that the impacts of mechanical excavation met a number of these criteria, and changes in fish abundance, habitat availability, sediment transport and water quality were all observed (Chapter 2, 3 & 5). These findings demonstrate that mechanical excavation of macrophytes is a potential source of disturbance in New Zealand waterways. They indicate that the effects of macrophyte control on fish populations (Mortensen, 1977; Swales, 1982; Serafy et al., 1994; Garner et al., 1996; Young et al., 2004) and their environment (Brookes, 1988; Wilcock et al., 1998; Young et al., 2004) may have been underestimated in the limited research previously undertaken.

There is increasing recognition that macrophyte removal is detrimental to freshwater fish populations (Swales, 1982; Hudson and Harding, 2004), which is supported by the results of the present study. Macrophyte excavation was found to reduce native fish abundance significantly in the short term (Chapter 2). These findings are consistent with international research (Swales, 1982; Serafy et al., 1994), and highlight the potential for disturbance during and after macrophyte removal. Fish population recovery after macrophytes were excavated was not monitored in this study, and the long-term impacts of macrophyte removal remain unclear. The effects of macrophyte removal on native fish communities may be partially offset by the life history patterns of many New Zealand species. Amphridromy, a type of diadromy, is common on oceanic islands such as New Zealand (McDowall, 2010), and the majority of the species caught before and after excavation in the Waituna catchment were amphidromous (Chapter 2). Adult amphidromous fish spawn in freshwater where the eggs develop. Upon hatching, larvae are transported to the ocean where they feed for a period of weeks to months. Developed juveniles then return to freshwater where they spend the remainder of their lives, thereby completing the cycle [summarised in

McDowall (2010)]. It is thought that this life history pattern enables fish to deal with the regular disturbances common to island riverine ecosystems by facilitating constant reinvasion of disturbed systems (Leathwick et al., 2008; McDowall, 2010; Crow et al., 2013). The ability of amphidromous species to reinvade after disturbances means that reduced fish abundance after macrophyte removal may only persist until displaced or killed individuals are replaced by recruitment during the next juvenile migration. Despite this, changes in demography within fish populations (skewed to younger, smaller fish) may have a significant impact on ecosystem function.

The role macrophyte removal has played in shaping the current distributions of native freshwater fish species needs to be quantified if the true long-term effects of this activity are to be understood. However, there is insufficient historical data, on both macrophyte removal and fish distributions, to run the statistical models required to achieve this. In addition, the almost universal use of excavation to control macrophytes in low altitude waterways (Hudson and Harding, 2004) means that isolating the effects of macrophyte removal history by partialling out the multitude of other factors known to influence fish distribution (e.g. distance from sea, altitude etc.) would be extremely difficult (D. Booker, NIWA pers. comm.). In the future, multi-catchment longitudinal before-after-control-impact studies could be conducted to quantify changes in fish community structure and demography following large macrophyte removal operations. Such work would provide insights into the recovery rates of lotic ecosystems after mechanical excavation of macrophytes, and would help determine the effects historical macrophyte removal has had on current fish distributions. In addition, this work could quantify the relative importance of nearby adult populations and juvenile migratory fish in the colonisation of recently excavated waterways.

6.2 Sediment as a driver of disturbance

6.2.1 Sediment resuspension caused by macrophyte control

At natural levels fine sediments are integral to the function of aquatic ecosystems, playing important roles in habitat heterogeneity, organic matter transport and bacterial and algal productivity (Droppo, 2001; Yarnell et al., 2006; Kemp et al., 2011). Sediment input into unaltered ecosystems is variable, and is dependent on season, catchment characteristics and the occurrence of catastrophic natural events such as landslides (Lazar et al., 2010; Kemp et al., 2011). With the global intensification of agriculture, forestry and mining, accompanied by accelerated urban development, fine sediment, once an important component of good water quality, has become a

form of pollution. Sediment is now considered to be one of the greatest threats facing riverine biodiversity (Waters, 1995; Henley et al., 2000)

The results reported in this thesis suggest that sedimentation may be particularly detrimental in low-altitude waterways utilised for the drainage of pastoral land. In dense macrophyte stands sediment storage is increased (DiToro, 2001; Waterman et al., 2011; Jones et al., 2012; Krevs and Kucinskiene, 2012), which raises the stream bed, exacerbating the influence macrophytes have on hydraulic capacity and increasing the need for their removal (Hudson and Harding, 2004). Large amounts of deposited sediment was resuspended during the macrophyte removal operation studied in this research [120,000 % increase in SS concentration (Chapter 3)]. Furthermore, without macrophytes to encourage deposition, SS concentrations remained elevated for more than two months after macrophyte removal. Until now, it has been generally thought that any increases in SS following macrophyte removal are temporary (Brookes, 1988; Waters, 1995; Wilcock et al., 1998; Young et al., 2004). The findings of this study refute this premise, and demonstrate that real-world macrophyte removal operations have a greater effect on sediment transport than has been previously predicted on the basis of small-scale experimentation (Brookes, 1988; Wilcock et al., 1998; Young et al., 2004).

6.2.2 The effects of sediment resuspension

The elements of this study provide evidence that sediment resuspension may be a source of disturbance in fish communities after macrophyte removal (Chapters 2, 3 & 5). The sediment concentrations recorded in the Waituna catchment after macrophytes were excavated (Chapter 3) were sufficient to induce avoidance behaviour in salmonids (Sigler et al., 1984; Scheurer et al., 2009) and juvenile native fishes (Boubée et al., 1997). Consequently, fish abundance may be reduced by sediment suspended during macrophyte removal, especially during juvenile migrations. SS has a multitude of direct and indirect negative effects on adult freshwater fish that could induce avoidance (Bruton, 1985; Henley et al., 2000; Kemp et al., 2011), the most important of which are outlined below:

1. Impaired feeding performance resulting from reduced reactive distance (Barrett et al., 1992; Hazelton and Grossman, 2009a);
2. Reduced food availability resulting from reduced primary productivity (Bruton, 1985; Van Nieuwenhuyse and LaPerriere, 1986; Davies-Colley et al., 1992) and invertebrate prey abundance (Quinn et al., 1992; Wood and Armitage, 1999);

3. Reduced respiration due to gill abrasion and clogging (Lake and Hinch, 1999; Sutherland and Meyer, 2007).

These results of Chapter 4 suggest that excavation increases SS to such an extent that the feeding rates of brown trout (*Salmo trutta*) are reduced. Quinn et al. (1992) reported that SS concentrations below those recorded in Chapter 3 are sufficient to reduce macroinvertebrate abundance. Therefore, the effects of reduced feeding performance on the health of brown trout (*Salmo trutta*) are likely to be exacerbated by reduced prey availability after macrophyte removal (Shaw and Richardson, 2001). Salmonids have been shown to leave high-sediment environments, thereby avoiding the negative effects of reduced feeding (e.g. reduced growth rates) (Sigler et al., 1984). If brown trout do leave the excavated reaches of streams, the reasons for their departure may include reduced prey capture success and lower food availability. Reduced feeding rates at sediment concentrations below those observed following macrophyte removal in Chapter 3 have also been recorded in the juvenile life stages of native fishes (Rowe and Dean, 1998). Therefore, impaired feeding may contribute to native fish avoidance of high SS concentrations (Boubée et al., 1997), and may reduce abundance after mechanical excavation of macrophytes. Future studies focused on quantifying the effects of SS on the feeding abilities of adult life stages of native species is required to increase understanding of the ecological effects of macrophyte removal.

SS concentrations recorded after a large macrophyte removal operation (Chapter 3) were not found to affect oxygen consumption rates of brown trout in this study (Chapter 4). However, it cannot be concluded that the respiratory performance of brown trout in recently excavated waterways is not affected by SS. In this study, fish were experimentally exposed to sediment for a very short time (90 minutes). Gill damage increases with length of exposure to sediment (Sutherland and Meyer, 2007). Consequently, the effects of long-term increases in SS concentration (Chapter 3) may have been underestimated in this study. Future studies focused on quantifying the effects of long-term sediment exposure on the respiratory performance of multiple fish species is needed to accurately determine the effects of macrophyte removal on fish health.

Sediment resuspension may alter fish communities after large macrophyte removal operations through avoidance (Chapter 3), but the short-term reductions in fish abundance observed in this study (Chapter 2) are more likely to be driven by associated lethal changes in water chemistry. Small-spatial-scale excavation experiments, like the one presented in Chapter 2, have been shown to have a minimal impact on SS in the long term (Wilcock et al., 1998; Young et al., 2004). Therefore, it is unlikely that sediment resuspension after experimental macrophyte removal

(Chapter 2) persisted for a sufficient period of time to influence fish health and abundance through the mechanisms tested in Chapter 4.

The results described in Chapter 5 demonstrate that water column de-oxygenation though oxidation of anaerobically-decomposed organic SS has the potential to reduce native fish abundance after macrophyte removal. In hypoxic conditions, fish are unable to absorb sufficient oxygen across the gills to meet energetic demands. Short-term exposure to dissolved oxygen (DO) saturations below 10 %, or persistent exposure to saturations below 30 %, is lethal to most of New Zealand's native fishes (Dean and Richardson, 1999; Landman et al., 2005). In this study statistically detectable increases in the occurrence and persistence of hypoxic conditions were observed after mechanical excavation of macrophytes in Waikato streams (Chapter 5). Although, periods of hypoxia were also observed after herbicide application in this study (Chapter 5), decreases in DO were gradual, and fish most likely had sufficient time to leave treated waterways before oxygen conditions became lethal. Conversely, the persistence of hypoxia immediately after excavation was such that significant mortality of native fishes could be expected (Landman et al., 2005). This is supported by the large numbers of visibly stressed giant kokopu observed after macrophyte removal in streams in the Waikato region and the catchment of Waituna Lagoon. These results are consistent with the effects of sediment resuspension that have been reported in other systems. Deoxygenation during sediment resuspension has been recorded in streams (Waterman et al., 2011), and has been found to kill large numbers of fish in lakes (Bruton, 1985). The results of this study (Chapter 5) suggest that changes in water chemistry associated with sediment resuspension is one of the greatest threats to fish health during excavation of macrophytes, and water column de-oxygenation may contribute to decreased abundance after macrophyte removal (Chapter 2).

Deposition of the sediment suspended during and after the macrophyte removal may have a significant impact on fish communities in downstream receiving environments. Deposited fine sediment affects fish directly by smothering developing eggs (Kemp et al., 2011); reducing oxygen availability near the substrate (Bruton, 1985; Henley et al., 2000) and altering habitat suitability and availability by infilling interstitial spaces (Walling and Amos, 1999; Collins and Walling, 2007); reducing the availability of plant cover (Yamada and Nakamura, 2002); and changing riffle-pool sequence structure (Berkman and Rabeni, 1987). The impact of deposited sediment on lower trophic levels may also affect downstream fish communities indirectly after macrophyte removal (Henley et al., 2000; Kemp et al., 2011). Deposited fine sediment reduces food and benthic habitat availability to invertebrates (Kemp et al., 2011) by smothering periphyton and macrophytes

(Brookes, 1986; Graham, 1990; Ryan, 1991; Yamada and Nakamura, 2002) and infilling interstitial spaces (Walton et al., 1977; Kemp et al., 2011). In addition sediment deposition can affect benthic invertebrates by reducing dissolved oxygen near the substrate (Crosthwaite et al., 2008; Sear et al., 2008). The effects of sediment deposition on macroinvertebrates can alter food availability to the fish species that prey upon them (Wood and Armitage, 1999; Matthaei et al., 2006), which will likely affect growth rates and community structure (Henley et al., 2000; Kemp et al., 2011). Future research focused on quantifying the effects of macrophyte removal on downstream sediment deposition is needed if the true effects of this activity on fish communities are to be understood and managed.

6.2.3 Managing the effects of sediment.

The results of this research highlight the need for future research to develop techniques for reducing sediment resuspension during macrophyte removal. Dense macrophyte stands act as sediment traps, encouraging the deposition and retention of suspended material (Jones et al., 2012). Consequently, leaving sections of waterway un-excavated may reduce downstream transport of suspended sediment during and after macrophyte removal. Future research could focus on determining whether the effects of large macrophyte removal operations on SS can be reduced without impeding drainage outfall, by maintaining sections of macrophyte beds or employing artificial sediment traps.

From a management perspective, it would also be extremely useful to be able to predict the potential for sediment resuspension in a system without prior experimentation or surveys. Sediment regime is dependent on a number of factors, and the effects of macrophyte removal will vary between catchments and waterways within catchments (Bruton, 1985; Lazar et al., 2010). Existing sediment regime models, like the Integrated Catchments Model for Sediments (Jarritt and Lawrence, 2006; Jarritt and Lawrence, 2007) require significant preliminary data collection (Lazar et al., 2010). Although these systems are certainly useful from a scientific perspective, they are too complicated for use in day-to-day on-the-ground management of regular macrophyte removal operations. A potential avenue for future research is the development of a simplified statistical model to quantify the influence of basic parameters (clearance history, morphology, catchment land use, hydrology, vegetation characteristics etc.) on the rates of sediment deposition in low-altitude New Zealand streams. The risk of sediment resuspension could then be easily predicted, and the need for mitigation assessed before macrophytes are excavated by considering the macrophyte removal history and the physical characteristics of a targeted system.

6.3 Other drivers of disturbance

6.3.1 Stranding

Although sediment resuspension after macrophyte removal undoubtedly has significant population-level effects on resident fish, stranding is likely to be another significant driver of disturbance. During excavation of macrophytes large numbers of fish are removed with the vegetation, and in some cases up to 20 % of the population can be displaced (Serafy et al., 1994). Without human intervention the majority of stranded fish die (Young et al., 2004). An original goal of this thesis was to isolate the factors that influence stranding during mechanical excavation of New Zealand waterways. Unfortunately, that component of this study could not be conducted because of the 2012/2013 drought in the Waikato region. Future studies should focus on identifying the manner in which factors such as macrophyte removal history, stream morphology and catchment use influence stranding rates during mechanical excavation. The information gained from a study such as this would make it possible to identify the systems that would benefit most from fish-recovery operations during macrophyte removal.

6.3.2 Habitat loss

Habitat loss after macrophyte removal is another potential driver of reduced fish abundance that requires further research. Macrophytes play an important role in increasing habitat complexity in streams, and are utilised by fish for cover and spawning habitat (Mortensen, 1977; Garner et al., 1996). Macrophytes also increase the availability of invertebrate prey for fish (Voigts, 1976; Gregg and Rose, 1985; Carpenter and Lodge, 1986; Kaenel et al., 1998; McAbendroth et al., 2005; Cortelezzi et al., 2013). Excavation of macrophytes removes almost all of the plant biomass from the targeted waterway (Kaenel et al., 1998). Homogenising the stream bed to such an extent undoubtedly reduces the number and diversity of fish the system can support (Hicks and Reeves, 1994). Although total macrophyte removal is likely to be detrimental to fish, the results of this study suggest that partial excavation may help mitigate some of the effects of this activity by providing refuges for species like the giant kokopu (*Galaxias argenteus*) (Chapter 2). Macrophyte biomass in New Zealand is seasonably variable, and decreases significantly during winter (Wilcock et al., 1999a). Therefore, it is not surprising that native fish are able to tolerate partial habitat loss. Future research should focus on developing cost-effective methods either for maintaining fish habitat during excavation (Chapter 2), or providing artificial habitat after macrophyte removal (Hicks and Reeves, 1994; Whiteway et al., 2010).

6.4 Conclusion

This study provides evidence that mechanical excavation of macrophytes reduces native fish abundance in New Zealand streams. Suspended sediment was significantly increased by macrophyte removal, and remained elevated for an extended period after mechanical excavation was completed. This finding contrasts with the results of other studies (Wilcock et al., 1998; Young et al., 2004). The results of this research suggest that sediment resuspension after macrophyte removal limits oxygen availability to resident fish, and reduces the feeding rates of brown trout. Until now the effects of macrophyte removal on fish community structure have been attributed mainly to habitat loss and stranding (James, 2013). This research provides evidence that sediment resuspension after macrophyte removal is also an important driver of disturbance in fish communities.

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