A small agricultural waterway in the upper South Island. Livestock, including sheep and horses, had direct accesses to the water and a lack of riparian planting resulted in low levels of shade.

THE IMPACT OF SUSPENDED AND DEPOSITED FINE INORGANIC SEDIMENT ON NEW ZEALAND FRESHWATER FISHES.

This thesis is submitted in partial fulfilment of the requirements for the Degree of Master of Science
by
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TABLE OF CONTENTS

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The fish sampling methods and laboratory experiments were carried out under ethic permit issued by the University of Canterbury Ethic panel. I would like to thank the Department of Conservation for approving 187 sampling sites and providing an electric fishing permit.

ABSTRACT

The introduction of excessive amounts of fine inorganic sediment into waterways is one major pathway by which anthropogenic activities exert negative pressure on freshwater ecosystems worldwide. Once in a waterway fine inorganic sediment particles either remain suspended (turbidity) or become deposited (sedimentation) depending on the water velocity and the particles size and shape.

A literature review revealed that both suspended and deposited fine inorganic sediment can negatively affected all trophic levels. Fish communities became dominated by sediment tolerant species. Moreover, these changes potentially reduce diversity and abundance in the top predators, thus diminishing food web length and decreasing ecosystem function. The combination of high levels of sedimentation and concomitant high turbidity, creates multiple stressors with the potential to work synergistically to further reduce diversity and abundance. Multiple fine inorganic sediment stressors imposed on key species in a trophic level potentially diminish energy flow to higher trophic levels, suggesting the most significant effects will be borne by the top predators.

I conducted a field survey of 52 stream and drains covering a sediment gradient, in the South Island of New Zealand. Results showed that several freshwater fish species were sensitive to deposited sediment particularly torrentfish, bluegill bully while several other species were not affected such as shortfin and longfin eels. Change-point analysis indicated marked declines in fish diversity and abundance between 5 - 25% deposited sediment cover.

Structural equation modelling indicated that both direct and indirect effects were responsible for sediment impacts on fish community composition. Sediment increases were tied to catchment land-use and fish diversity was linked to habitat size. The survey results support the hypothesis that fine inorganic sediment deposited on the bed of a waterway causes environmental stressors by homogenising habitat and reducing resource (i.e., invertebrates) quality and quantity.

To better understand if turbidity was responsible for changes in fish diversity or abundance, I investigated the effect of high turbidity on fish feeding response by conducting a series of 24 hour laboratory feeding trials. However the results did not fully support my hypothesis that high turbidity would reducing feeding success. Two (i.e., brown trout and shortfin eels) of the four species tested did not show a significant altered feeding response even when turbidity was as high as 200 NTU. Only the redfin bully (*Gobiomorphous huttonii,* Ogliby), showed any sensitivity to high turbidities. The Canterbury galaxias (*Galaxias vulgaris,* Stokell*)* data provided an unexpected significant result. As turbidity increased Canterbury galaxias feeding rate also increased. These results point to varied evolved tolerances to suspended sediment.

Overall, the results have provided new insights into the effects of fine inorganic sediment on New Zealand freshwater fish communities and provide some guidelines for waterway management.

PREFACE

This thesis addresses two gaps in our current knowledge. Firstly, what are the effects of fine in-organic bed sediment on freshwater fish community composition in New Zealand? I investigated the potential influence of sedimentation on multiple levels of biological organisation (i.e., species and communities). I postulated that since bed sediment cover exceeding 20 % altered invertebrate community composition that the native fish which feed on invertebrates would also be impacted by changes in food resources and by homogenisation of benthic habitat.

Secondly, do New Zealand fish display thresholds at which turbidity negatively effects feeding success rates? I hypothesised that high levels of turbidity would reduce the ability of fish which are often seen as visual predators to locate prey. I expected our fishes would have varying levels of tolerance to both bed and suspended sediment.

In the Chapter 1 I synthesised the available peer-reviewed literature and discussed some relevant current theory. Moreover, I identified apparent gaps in our knowledge and developing several hypothesis to test.

Chapter 2 I was able to determine that deposited sediment had negative effects on the taxonomic richness of fish communities and tested the mechanism that drove those changes.

Chapter 3 covers provides details of the threshold responses to sediment deposition by individual fish tax and the boarder community. In Chapters 2 and 3 the significant effects of deposited sediment were detailed, raising the question does entrained sediment impact fish? To better understand the effects of turbidity, I undertook 24hr day / night laboratory feeding trials with four species (i.e., shortfin eels, brown trout, Canterbury galaxias and redfin bully) at five different turbidity levels and the results are presented in Chapter 4. This final chapter (Chapter 5) summarises the previous chapters and adds further discussion where appropriate, whilst providing some suggestions regarding the management implications of sedimentation in waterways under the current climate change predictions.

The fish sampling methods and laboratory experiments were carried out under an ethic permit issued by the University of Canterbury Ethics Panel. The Department of Conservation approved 187 sampling sites and provided an electric fishing permit.

CHAPTER 1

AN INTRODUCTION TO THE EFFECTS OF EXCESSIVE FINE INORGANIC SEDIMENT IN LOTIC ENVIRONMENTS

A stream in a National Park on the West Coast of the South Island of New Zealand. The bed substrate consists of pebbles, cobbles and boulders with few inorganic particles < 2 mm, therefore is described as low sediment (5% bed cover). This waterway is representative of a pre-colonisation state when the bed sediment average cover estimated at only 8% and high quality riparian margins were the norm.

OVERVIEW

The Anthropocene era has seen freshwater ecosystems become the most imperilled biomes on earth (Vitousek et al. 1997, Sala et al. 2000, Dudgeon 2010), facing degradation from intensive and extensive land-use changes (MacLeod and Moller 2006). New Zealand is no exception, when Māori arrived in the 13th century they cleared forests and established agriculture (Ogden et al. 2006). While in the last 200 years European colonisation has led to further widespread land clearance and intensification of land-use. Today, farms, exotic forests, industrial parks and urban centres, have replaced podocarp, beech and mixed broadleaf forests, indigenous grasslands and wetlands (Ewers et al. 2006, MacLeod and Moller 2006). Land-use changes have already resulted in radically modified channel geomorphology, altered flow regimes (Newson 2000) and increased sediment (defined here as inorganic particles of less than 2 mm in diameter) inputs (Walling 1983). Human population growth, combined with the expectation that more resources will be available to each successive generation (Holdren and Ehrlich 1974), will necessitate further land conversions and land-use intensification.

Sediment occurs naturally in waterways, however anthropogenically driven sources have increased since the colonisation of New Zealand (Clapcott et al. 2011). Run off from many land-uses including agriculture, urbanisation, silviculture, road construction and extractive industries (e.g., mining and quarrying), enters waterways carrying fine sediment (Davies-Colley et al. 1992, Death et al. 2003, Davie 2004, Walling 2005). Furthermore, poor riparian management, removal of riparian vegetation and narrow riparian buffers can result in bank failure and erosion, directly increasing sediment loads in waterways. When livestock, in particular sheep and cattle, have direct access to waterways, pugging and erosion further escalates direct sediment input (Holmes et al. 2016). Furthermore, non-point source inputs from agricultural tile drains, and open drains can be additional sources of sediment (Neal et al. 2008) which more difficult to control due to their discrete nature.

Arguably, New Zealand's urban waterways are in the worst condition with, legacies of sediment from poor riparian management, hard-surface run-off and inappropriate wastewater discharge (Wilson 1989, McEwan and Joy 2009, Zygadlo 2016). Over decades, altered flow regimes, including reduced flows (due to poor governance of water abstraction) (Memon 1997), combined with increased sediment inputs (Walling 1983) have resulted in a build-up of deposited sediment, creating a sediment legacy (Fryirs 2013).

Sediment only becomes a biotic stressor when it exceeds a species or communities evolved threshold. As an anthropogenically driven stressor, excessive sediment represent an evolutionarily novel condition for biota, one which falls outside the range of conditions experienced by extant species. Freshwater biota evolved under specific naturally occurring levels of deposited sediment and turbidity concentrations. Historically, changes have occurred over geologic time periods. When sediment norms are exceeded by anthropogenically driven inputs, they are exceeded at an accelerated pace (Fox 2007, Williams and Jackson 2007). For multicellular organisms, this accelerated pace of change almost ubiquitously exceeds the ability of species to adapt (Hobbs et al. 2006, Fox 2007, Williams and Jackson 2007). I believe it is under these novel conditions that sediment becomes a ‘negative subsidy’, (a donor-mediated resource which has a negative impact), causing biotic stress and leading to altering community composition at multiple trophic levels.

High sedimentation and turbidity can decrease algal productivity (Davies-Colley et al. 1992, Luce et al. 2013) by smothering algal communities, reducing light for photosynthesis and abrading foliage. Furthermore, sedimentation alters other basal resources by burying detrital material and modifying microbial activity (Danger et al. 2012) and taxonomic composition.

At higher trophic levels benthic invertebrate communities changed in response to sedimentation. When bed sediment exceeded 20% in Canterbury agricultural waterways, communities became less specious and were dominated by molluscs and burrowing aquatic (Burdon et al. 2013). Reduced or altered diversity at one trophic level, can influence the next trophic level via diminished transfers of energy and thereby disrupting ecosystem function.

Apex species often face the most serious impacts; in New Zealand the apex predators in freshwater systems are the fish.

A REMARKABLE FISH COMMUNITY

At present there are 50 extant New Zealand native freshwater fishes recognised (Appendix A), with several species yet to be formally described. There are an additional three species considered colonists and one extinct species. (Allibone et al. 2010, McDowall 2010, Goodman et al. 2014). This represents a relatively depauperate community when compared with other similarly sized temperate island nations. Japans freshwater community comprises 96 genera, with 188 native species and subspecies (Yuma et al. 1998). Although depauperate, the New Zealand fauna is uniquely endemic, with more than 30 species found nowhere else in the world, and as yet unrecognised species also likely to be considered endemic (McDowall 1990, Allibone et al. 2010, Goodman et al. 2014). This high endemism is a result of New Zealand’s long post-Gondwana isolation, perhaps as long as 80 million years (McDowall 2010, Goodman et al. 2014). More than half of all native fish taxa belong to the family Galaxiidae. The remainder belonging to only eight other families (i.e., Geotriidae, Anguillidae, Retropinnidae, Prototroctidae, Cheimarrhichthyidae, Eleotridae, Mugilidae, and Pleuronectidae) (McDowall 1990), suggesting adaptive radiation (McDowall 2010).

In addition to the native fish species, 20 exotic species have been introduced to New Zealand, mainly as sport fish, unsuccessful biological controls or via the aquarium trade (McDowall 1990). When

Europeans colonised New Zealand the waterways appeared largely uninhabited by fish. This was due to the cryptic nature and secretive nocturnal behaviours of native fishes. For example, the banded kōkopu (*Galaxias fasciatus*, Gray) uses undercut banks, boulders or woody debris to remain undetected (Rowe and Smith 2003). The perceived lack of fish and the colonists desire to re-create British ecology in New Zealand, led to stocking of the waterways with European species. In 1867 the Acclimatisation Society introduced brown trout (*Salmo trutta*, Linnaeus) which have established successfully (Kos 2013) along with two other Salmonidae taxa, rainbow trout (*Oncorhynchus mykiss*, Walbaum) and Chinook salmon (*Oncorhynchus tshawytscha*, Walbaum) (McDowall 1990).

When the Acclimatisation Society released brown trout in the 1860's, they also introduced a Salmonidae parasite, (*Myxobolus cerebralis,* Hofer). Currently found only in South Island brown trout populations, it causes whirling disease, leading to spinal malformations. *M. cerebralis* requires a oligochaete (*Tubifex tubifex*, Muller) to complete its life cycle (Brinkhurst 1996). No occurrences of *M. cerebralis* have been documented in native species thus far. Nevertheless, its presence does provide opportunity for cross-species infection (Baerwald et al. 2011). In Europe infections have occurred in rainbow trout and the United States, bull trout (*Salvelinus confluentus,* Suckley), cutthroat trout (*Oncorhynchus clarkia,* Richardson) and rainbow trout have been infected. Notably all species infected to date have been in the family Salmonidae (Bartholomew et al. 2003, Koel et al. 2006).

Introductions did not stop with salmonids, there are several Cyprinids (cyprinidae), however only goldfish (*Carasius auratus*, Linnaeus), rudd (*Scardinius erythrophthalmus*, Linnaeus) and tench (*Tinca tinca*, Linnaeus) are found in the South Island, where they primarily inhabit lentic environments. The Poecillidae were introduced mainly as tropical aquarium species and are not established in the South Island. Perch (*Perca fluviatilis,* Linnaeus) are well established in both lentic and lotic environs throughout the South Island, although they prefer lakes and slow moving waters. As they are piscivores they reduce the abundance of common bully (*Gobiomorphus cotidianus,* McDowell), īnanga (*Galaxias maculatus,* Jenyns), smelt (*Retropinna retropinna*, Richardson) and Southern freshwater crayfish (*Paranephrops zealandicus,* White) in lakes, suggesting their presence in rivers presents a similar threat to native species.

In New Zealand there are 3,820 lakes with a surface area greater than one hectare (New Zealand. Ministry for Culture and Heritage 2003). Despite the size of available lentic habitat, the majority of freshwater fishes are riverine, being found almost exclusively in lotic habitats. A few taxa are found in both rivers and lakes including the longfin (*Anguilla dieffenbachia,* Gray) and shortfin eels (*Anguilla australis,* Richardson), common bully, īnanga and giant kōkopu (*Galaxias argenteus,* Gmelin). Conversely no species are found exclusively in lentic environments (McDowall 1990).

New Zealand's native fish community is not only depauperate and uniquely endemic, but also consists of species that are mostly small (less than 150 mm) and benthic rather than pelagic (McDowall 1990). Even more unusually, some species spend the majority of time within the substrate, living below the stream bed in interstitial spaces (McEwan and Joy 2014b). However, there are exceptions such as īnanga a pelagic species commonly found inhabiting the upper reaches of the water column (Sagar 1993, McDowall 2000). The short-jaw kōkopu (*Galaxias postvectis*, Clarke) requires different diel habitats, being found in areas with larger substrates during the day, than those at night (McEwan and Joy 2014a). Despite their benthic nature most New Zealand fish foraging for prey in the water column.

An ability to detect prey drifting in the water column is important as nearly all New Zealand native fish are predatory, consuming invertebrates and smaller fish. Interestingly, the community contains no native herbivorous or planktivorous. Although the lamprey (*Geotria australis,* Gray) is specialized to feed on detritus when in freshwater, thus in-habitats a totally different trophic niche to other native fishes (James 2008). Lamprey larvae, called ammocoetes, spend about four years as filter feeders buried in fine sediments. Then they develop into tiny adults (James 2008) and migrate to the ocean.

Over half of New Zealand fishes have marine tolerant life stages, again, an unusually high proportion when compared with fish communities worldwide. In North America, from a total fish community of 1,200 taxa, only 24 are diadromous, (0.02%) (Limburg and Waldman 2009). Diadromy involves movements, between marine and fresh waters necessitating physiological changes (McDowall 1990, 1993). There are three very distinct forms of diadromy, each represented by species present in New Zealand. In amphidromous migrations fishes live, breed and deposit eggs in freshwater, their larvae drift to the ocean to feed and develop, returning to freshwater as fry. For example, the New Zealand whitebait industry is based on harvesting the fry of five amphidromous species as they return from the marine environment to freshwaters. However, in anadromous migrations, feeding and growth occur at sea prior to the adult fish returning to freshwater to reproduce (e.g. some introduced salmonids). New Zealand's two species of eels are known to be catadromous, their feeding and growth occurs in freshwater prior to migration of the adult fish to the marine environment to reproduce. New Zealand has an unusually high proportion of amphidromous or catadromous taxa. Conversely, in most migratory community anadromy dominates while amphidromy is almost absent (McDowall 1998). McDowall, (1996) highlighted that the high proportion of diadromous species' in the New Zealand fauna was important to help understand distribution. Moreover, distribution is further determined, by a diadromous taxa's ability to penetrate inland, an important factor to consider when sampling fish communities.

Fish diversity is almost universally found to increase when moving in a downstream direction from the headwaters. This pattern has generally been attributed to increased habitat diversity and volume, and more stable conditions in the lower reaches of a waterway. However, the same downstream increases in diversity in New Zealand, have been attributed the dominance of diadromous fauna (McDowall 1993, Jowett et al. 1996, McDowall 1998).

The distribution and upstream penetration of diadromous fishes is influenced by a species' migratory ability, its instinctive drive and the physical attributes of the waterway (McDowall 1990, 1993). Impediments to migration within a river may be natural (e.g., waterfalls, high flows or closure of the mouth) (McDowall, 1993) or anthropogenically derived, (e.g., dams and contaminants including high turbidity or low pH) (McDowall 1990, Boubée et al. 1997).

Diadromy provides opportunity for dispersal into new habitats and allows juvenile fish to 'select' a waterway upon return (McDowall 1998). River selection by migratory juveniles is known to rely on pheromonal cues from adults (Baker and Hicks 2003), but even moderately turbid water has been shown to lead to avoidance by some taxa (McDowall 1996, Boubée et al. 1997). Turbid river mouths could inhibit the migratory colonisation of headwater streams by juvenile fish, despite the quality of habitat and water further upstream.

Globally nocturnal species are known to use non-visual senses (e.g., olfactory, social, electrical and mechanosensory), in combination with visual senses to detect and avoid predators and locate prey. New Zealand has many nocturnal fish taxa, with several species known to use non-visual senses. As an example, torrentfish (*Cheimarrichthys fosteri,* Haast) have many superficial neuromasts and a simple unbranched canal system which forms a relatively well developed mechanosensory system; a development likely in response to nocturnality (Carton and Montgomery 2004). Jellyman and Todd (1989) noted that eels feed actively in turbid waters, therefore, feeding is not solely dependent upon visual senses. In both species of eels, olfaction is probably the main sense involved in the detection of food, while the specific location could be determined by any combination of senses (e.g., touch, taste, olfactory, visual, and movement detection) (Jellyman and Todd 1998).

Galaxiidae all have a primary trunk lateral line and well-developed sensory pores in the head (Cadwallader 1973). Some of the Galaxiidae family namely the *Neochanna* (commonly known as mudfish), have an enhanced olfactory sense (McDowall 1997). Like eels, they have elongated, tubular, and forward-facing anterior nostrils. Using olfactory senses in conjunction with an extensive system of sensory pores, similar to those used by torrentfish, Neochanna are well-adapted to forage under low light conditions (Cadwallader 1975).

A VALUABLE COMMUNITY

Many New Zealand native fish are valued by Māori as mahinga kai, (McDowall 1990, Townsend et al. 2004) and also have recreational, commercial, ecological and conservation importance. For example, shortfin eel, longfin eel, lamprey and the fry of five diadromous Galaxiids, (that comprises the New Zealand whitebait catch), are all highly valued culturally, recreationally and commercially. Of these species, the longfin eel and two of the whitebait species are considered under pressure from harvesting (McDowall 1991, Environment 2014). Many Galaxiids and the redfin bully (*Gobiomorphous huttonii,* Ogliby*)* are designated as "*in decline, nationally vulnerable or nationally threatened*" (Goodman et al. 2014). As the majority of New Zealand native fish are endemic and threatened (McDowall 2010), they are considered of high conservation status, regardless of this status harvesting continues (Appendix A).

Despite only one species, the New Zealand grayling (*Prototroctes oxyrhynchus,* Günther), becoming extinct, New Zealand has one of the world's worst records for threatened freshwater fish species with 74% of all native species being listed as threatened or at risk (i.e., nationally critical, nationally endangered, declining), some 40 taxa (Goodman et al. 2014). The range and abundance of most species has declined since European colonisation (McDowall 1990, Goodman et al. 2014). Even the sport fishing industry, specifically the brown trout fishery, is seen by local anglers to be at risk. Both catch-sizes and numbers are dropping and in some previously well-known trout fishing areas such as the Selwyn River, fish are also most non-existent (Fishing.com 2017).

Also considered in 'gradual decline' by the 2012 New Zealand Threat Classification System are the two species of freshwater crayfish (Whitmore et al. 2000, Grainger et al. 2013). The Southern crayfish, found on the East coast, and the Northern crayfish (*Paranephrops planifrons,* White) found in the North Island and on the west coast of the South Island are both culturally important. They are known as kōura or kēkēwai by Māori who have traditionally harvested both species. Crayfish are important for ecosystem function as they are involved in bioturbation, significantly reducing sedimentation (Parkyn et al. 1997).

THE NEW ZEALAND GEOLOGICAL SETTING – UP, UP AND AWAY

New Zealand is a slender archipelago of islands, which lays on the boundary between the Pacific and the Indo-Australian plates and is part of the Pacific Ring of Fire (Coates and Cox 2002). The country's North Island, is home to Taupo, a mega volcano which has produced two of the world's most powerful eruptions in geologically recent times (27,000 and 254,000 years ago) (Bailey and Carr 1994). In the South Island's East coast the eroded remnants of two stratovolcanoes (Lyttleton and Akaroa) are prominent features. While further south, the Waiareka-Deborah volcanic group near Oamaru and the Dunedin Volcanic Complex dominate the coast line (Thornton 2003).

Earthquakes are common through-out the country, however, the Hikurangi-Kermadec subduction and alpine fault produce the most substantial activity. Tectonic plate movement along the Alpine Fault is uplifting the Southern Alps at 10 mm or more a year, making it one of the most rapidly rising mountain ranges in the world (Coates and Cox 2002). The uplifted rocks of the Alps, are being weathered and eroded by tectonic stresses, variable temperatures and high rainfall. Evidence of these processes is widespread, cones of eroding scree and expanses of boulders and gravel make up the beds of the rivers. The Canterbury Plains, another major feature of the South Island, were formed by these processes, as rivers carrying eroded debris to the coast deposit their loads on the plain (Thornton 2003).

The Southern Alps and Kaikōura ranges were both formed from 100–300 million-year-old sedimentary greywacke an ancient mudstone. In the north-west Nelson ranges there is an area of alpine karst (limestone and marble), however, much of the rock featured is granite and gneiss (Thornton 2003). In Banks Peninsular and Oamaru the prominent rock type is volcanic.

It is not only rock composition that is dictated by geology, New Zealand's complex climate and weather patterns are a result to the geology of the islands (Coates and Cox 2002, Thornton 2003). The climate varies widely from warm subtropical climes in the far north to cold temperate in the far south. The Southern Alps provide a barrier to the moisture-laden westerly winds, upon reaching the mountains the air rises, dropping rain on the West Coast. On the Eastern side of the Alps the winds become Nor-westerly, a hot dry wind that races across the plains. This results in the West Coast having the highest rainfall and being the wettest area of New Zealand. Conversely, Canterbury to the east of the mountains, is the driest (Statistics New Zealand 2017).

The ongoing uplift and climate driven weathering result in masses of material being eroded from the landscape annually, much of it in the form of sediment (Coates and Cox 2002).

SMALL PARTICLE MECHANICS

The term sediment refers to fine inorganic material less < 2 mm in size, encompassing, sand (< 2000 to > 62 µm), silt (< 62 to > 4 µm) and clay < 4 µm) (Appendix B).

Sediment supply

The natural supply of sediment is controlled by catchment, geology, topography, vegetation type and cover (Stromberg et al. 2009), rainfall, and catastrophic events, such as volcanic eruptions, earthquakes or extreme storms (Hicks and Griffiths 1992) and fires (Beaty 2011). While erosion is a natural process, the rate at which erosion is occurring globally has increased by 10 - 40 times driven by anthropogenic activity.

Sediment is introduced into freshwater systems by natural processes including weathering, in-stream abrasion, erosion and aeolian sources. Overland flow can erode soil particles and transport them downslope and into waterways. Aeolian sources, especially on the Canterbury Plains, where the Nor-west wind is strong, can deliver particles directly into waterways (Fig. 1.1).

Anthropogenic driven supply can come from a variety of sources (e.g., wastewater, tile drains and hard-surface run-off.)

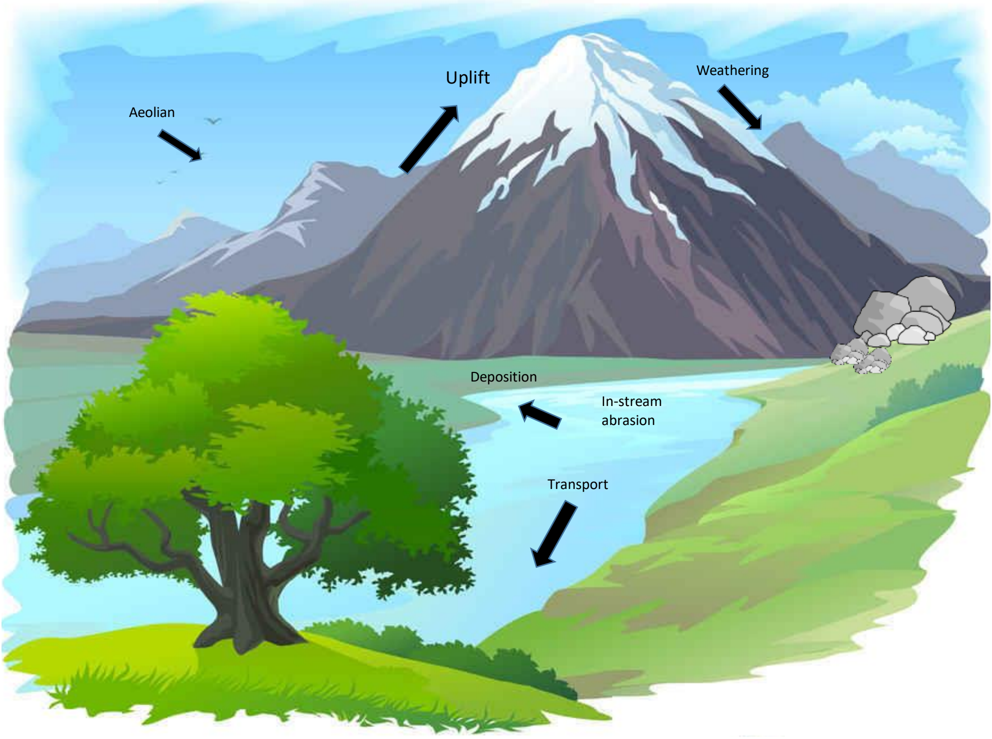

Figure 1.1 In natural geological cycles, orogeny and uplifting exposes rock to weathering. Errored rocks are worn by natural processes and can be transported into waterways by surface flows. Once entrained areas of water velocity allow particles can be deposited, often to be uplifted at some future time. As the abrasive process occurs material gets progressively smaller, (e.g., boulders become cobbles) and eventually forms sediments (i.e., sand, silt and clay). Grain sizes are given in the Wentworth scale panel (right), sediment covers clay silt and sand.

Type	Size
Large boulder	>1024
Medium boulder	513 - 1024
Small boulder	257 - 512
Large cobble	129 - 256
Medium cobble	91 - 128
Small cobble	65 - 90
Coarse gravel	17 - 64
Medium gravel	9 - 16
Fine gravel	2- 8
Sand	< 2000 > 62 µm
Silt	< 62 to > 4 µm
Clay	< 4 µm

Sediment transport

After entering the waterways particles can be transported suspended anywhere within the water column, depending on the difference between the particles shear stress and settling rate. Settling rate, describes the tendency for particles to fall out of suspension; boundary layer shear stress describes the tendency for moving fluid to bring sediment particles into suspension (Fig. 1.2).

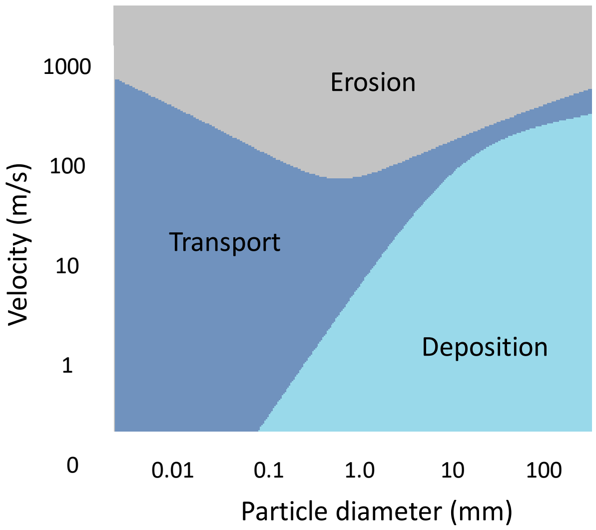

Figure 1.2 Whether sediment will be eroded, transported or deposited is depended on the particle size and the flow rate of the water.

Stokes settling defines the velocity of a spherical particle settling through liquid, however, for very small particles the Reynolds number must first be derived (refer Appendix C). Stokes calculates the difference between, drag force (which slows the particle down), and gravity (which acts to speed the particle up). If these two forces are balanced, the particle is not accelerating (Batchelor 1967).

When a particles settling velocity is more than the upwards velocity but it has sufficient momentum to keep moving forward it is defined as bed load. This sediment is not truly suspended, and its movement is not uniform nor continuous instead particles roll, slide, bounce and saltate (from Latin saltus, "leap") downstream. When a particles settling velocity approximately equals upwards velocity, it becomes suspended and is referred to as suspended load. An important constituent of the suspended load is wash load, which is comprised of the finest suspended sediment, typically less than 0.00195 mm in diameter. The wash load is differentiated from the suspended load because it will not become deposited during low flow events. Turbidity in slow-moving waterways is attributed to wash load. For example, in meltwater rivers glacial flour (suspended clays < 4 µm) forms part of the wash load and remaining entrained even in low velocity reaches.

Turbidity

Turbidity refers to the propensity of particles (organic and /or inorganic) to scatter light. Scattering contributes to attenuated light, because it forces light to take a convoluted path through the water column, increasing the probability of it being absorbed. The effect of entrained sediment particles on light is used as a turbidity measure, referred to as Nephelometric Turbidity Units (NTU).

Measurements of suspended sediments are typically reported in either turbidity levels (NTU) or suspended sediment concentrations (mg/L) (Davies-Colley et al. 1993).

The depositional process

When the energy available for sediment transport is not sufficient to overcome gravity and friction, particles become deposited, called the null-point. Rates of deposition depend on both the physical characteristics of the particles, (e.g., spherosity, size), water velocity and turbulence (determined by morphology and discharge). Because deposition requires a reduction in energy (particles rarely change size or shape significantly), sediment is not deposited evenly over the bed of a waterway (e.g., in a point bar, the outside of a meander or alluvial fans). The null-point hypothesis also explains how sediment is deposited according to its grain size.

However, it is the proportion of suspended load to bed load that is important for channel morphology.

WATERSCAPES FROM GEOLOGY

New Zealand has an abundance of waterways of various forms (e.g., braided, meandering, glacial and spring fed) and with highly varied substrate compositions (e.g., greywacke, gneiss and marble), which are often highly localised (Thornton 2003). A consequence of the country's narrow shape and uplifted costal Alps is short, steep, west-flowing rivers on the West Coast of the South Island when compared with those on the East Coast (Coates and Cox 2002).

West Coast - tannin streams

Many West Coast waterways are notable for their unusual tea-like shades of golden brown, a result of natural organic tannins released into the water from the decomposition of podocarp leaves. The tannins eventually leech into the waterways, naturally increasing turbidity and decreasing the pH which can be as low as 3 (Harding et al. 2004).

Despite high turbidity from organic material, suspended inorganic sediment loads are low; especially in catchments with limited anthropogenic impact. Similarly, deposited fine inorganic sediment cover is usually low, mainly as a result of low inorganic load and high water velocity, ensuring any particles present remain entrained.

West coast glacier-fed waterways

A feature of the mountainous West Coast is the glacier-fed waterways. New Zealand's glaciers are extremely close to the coast. This results in short, cold, highly turbid, high velocity waterways, flowing down moraine valleys, often over terrain un-affected by land-use changes. The exception being streams in areas were gold or coal is extracted, which are highly impacted. The turbid melt waters, are often milky in appearance, due to high concentrations of very fine inorganic suspended clays < 4 µm (glacial flour). The high water velocity and high wash load results in low sediment deposition.

Lowland spring fed systems

On the Canterbury Plains, Southland, Nelson and in some places on the West Coast, waterways fed by springs appear in the lowlands. These springs are driven by aquifers and often contain ancient groundwater. Groundwater can travel as little as one meter in a decade. On the Canterbury Plains where springs are common, rock-water contact purifies the water and ensures it is a steady temperature. Overall, spring habitats, tend to be stable, unless anthropogenic activities have had an effect on the aquifer (e.g. water quantity and quality) or the spring (e.g. water abstraction and riparian habitat modification). Numerous spring-fed lowland waterways are now surrounded by agricultural, horticultural or urban catchments, where they are often straightened and controlled, to ensure they function as conducts for water abstraction and flood control. In agricultural catchments their status as streams, rivers or creeks, is often stripped, instead they are relegated to the title of agricultural drains. I have endeavoured to redeem their status by calling all lotic systems, waterways where applicable.

Increased sediment loads, result in an increase in the abundance of invasive macrophytes. During spring and summer, lowland agricultural waterways are clogged with macrophytes that reduce water velocity, this in turn affects the deposition and conveyance of sediment, increasing retention of fine particulate material on the river bed (Jones et al. 2012) (feedback loop) and reduces the waterways function as a conduit for flood waters.

Clearing the macrophytes, with either herbicide sprays or mechanically, disrupts fish and other biota (Greer et al. 2015). Mechanical diggers scrape the bed of the stream uplifting macrophytes from the bed and thereby releasing plumes of suspended sediment downstream. These sediment pulses are known to reduce the feeding rates of brown trout by 22% when they exceed 450 mg L^{-1} (Greer et al. 2015). The macrophyte laden debris is deposited on the bank; unfortunately this often contains live fish and invertebrates, many of which are unable to make their way back to the stream (Beentjes et al. 2005, Greer et al. 2012). Herbicide sprays, do not disturb the bed, instead they leave the decaying macrophytes in the waterway. The resultant eutrophication leads to reductions in dissolved oxygen concentrations, potentially leading to fish deaths or immigration.

East coast - braided rivers

A distinctive feature of New Zealand's waterways, is its multiply-braided rivers, most of which cross the eastern plains (Canterbury Plains) of the South Island (e.g., Waipara River). A braided river is one that, over some part of its length, flows in multiple, mobile channels across a gravel floodplain. Braided rivers occur where there is steeply sloping terrain and/or high sediment loads. Although high bed load is associated with high turbidity, high velocity results in limited deposition of sediment usually only at point bars.

Slope and load are actually synonymous, an increase in sediment load will over time increase the slope of the river bed. In flume experiments increased slope and sediment load altered channel patterns significantly (Schumm and Khan 1972). Reductions in sediment loads within braided river systems eventually led to their transformation to meandering systems.

Meandering soft-bottomed waterways

In the North Island the terrain is not dominated by Alps such as it is in the South Island, instead volcanic cones are interspersed with rolling hills or flat plains. This changes both weathering and erosional processes and gives rise to naturally slow-moving meandering soft-bottomed waterways. Soft-bottomed waterways are less common in the South Island. These waterways often prove to have relatively high levels of turbidity (>10,000 NTU), especially after heavy rain (Rowe et al. 2002).

Extremely rare, but present in some parts of the North Island, are meandering waterways with beds of eroded ignimbrite. This lightweight material, which includes pumice, is ejected from volcanoes. Once transported into waterways it forms highly unstable beds, with relatively soft and easily eroded material. This material transports readily, even in low flow conditions. Due to its volcanic nature particles can be non-spherical and highly abrasive, increasing the impact on aquatic biota.

Nelson waterways

The Nelson region is known for its golden sandy beaches, which are derived from the regions unusual geology. At Red Hill in the Richmond Range, an outcrop of deep red ultramafic rocks, is rich in minerals containing iron and magnesium. Ultramafic rocks form poor soils and are readily eroded. The same soft golden sediments can be seen in some Nelson waterways (e.g., Kaiteriteri stream) where particles >2mm cover the majority of the stream beds (Fig. 1.3).

The importance of sediment

Unlike most pollutants, sediment is necessary for waterway function with ecosystems benefiting from both transport and deposition. In the Mekong and in some rivers in Europe, there is a focus on bedload, since virtually all bedload material is impounded by dams (Kondolf et al. 2014). Reductions in bedload results in sediment starvation (Wohl and Ebrary 2004). Sediment builds aquatic habitats for spawning and benthic organisms. Furthermore, sediment is responsible for providing nutrients to aquatic plants, and as we have already seen, it is responsible for waterway morphology, making it an integral part of the ecosystem.

In an example from the New Zealand sediment literature, Kelso & Todd (1993) concluded from research in two New Zealand streams, that sediment depth and composition were the dominant environmental factors influencing density and distribution of ammocoetes, (the larvae of the New Zealand lamprey). More recently, Jellyman & Glova (2002) investigated habitat used by ammocoetes in the Mataura River, Southland. They found high densities of ammocoetes were associated with run habitat and substrate of <1 mm in size, suggesting a sediment preference. Therefore, even in pristine ecosystems some sediment is necessary (Kelso and Glova 1993, Jellyman and Glova 2002), Despite the importance of fine inorganic sediment in freshwater systems, when excessive loads occur the negative impacts of sediment can be wide ranging.

THE CHANGING STATE – SEDIMENT IN NEW ZEALAND

One of the major diffuse pollutants effecting waterways worldwide is fine inorganic sediment, however, until the last decade when a paradigm shift occurred sediment affects were not recognised as a major issue. In 2011, Clapcott et al. modelled the pre-colonisation reference state as 8% instream sediment bed cover, implying hard-bottomed streams dominated the New Zealand waterscape. However the model predicted the national average in 2011 to be 29% (Clapcott et al. 2011); a significant increase, potentially due to anthropogenically driven land-use changes.

New Zealand's worst water quality, nationally, is found in urban or mine-affected rivers in which other contaminants such as toxic metals are elevated (Cottam and James 2003, Harding et al. 2004). Fortunately, these impacted waterways do not predominate the country. However, moderately impacted waterways do predominate over pristine waterways. For example, cropping (on 1.5% of the land area) and agriculture (54.8% of the land area) (Statistics New Zealand 2017) mobilise nutrients and sediment. Overall, water quality in waterways in exotic forests (7% of the land area) approaches the quality of rivers in native vegetation (Quinn et al. 1997). However, despite the good water quality fine sediment remains an issue in exotic forests (Quinn et al. 1997, Thompson and Townsend 2004). Urbanisation uses 1.9% of New Zealand's land (Statistics New Zealand 2017), indicating in total more than 60% New Zealand is being used for activates that negatively impact water quality and introduce sediment. Sediment has been termed a 'universal pollutant' due to its ability to impact ecosystems at multiple trophic levels and in two very different forms, suspended (turbidity) and deposited (bed).

DEPOSITED SEDIMENT - THE IMPACTS ACROSS MULTIPLE TROPHIC LEVELS

The most pervasive and wide ranging effects of fine inorganic sediment deposition comes through the altered physical structure of the bed. A loss of habitat complexity reduces refugia from predation and flow (Wood and Armitage 1997). However, sediment affects aquatic ecosystems through multiple pathways and at multiple topic levels (Culp et al. 1986, Berkman and Rabeni 1987, Angradi 1999).

Deposited sediment and basal resources

Deposited sediment impairs primary production (Wood and Armitage 1997, Dickman et al. 2005) and detrital breakdown (Danger et al. 2012). Wood and Armitage (1997), showed algal biomass decreased as a result of a combination of habitat loss and smothering of algal communities by sediment. Furthermore, once substrate interstices were filled sediment accumulated disconnecting the hyperetic zone from surface water (Nogaro et al. 2010). This disconnection reduced the amount of dissolved oxygen in the hyperetic zone, conditions, that have been shown to alter microbial activity (Nocker et al. 2007, Newman 2011) and change the taxonomic composition of the epilithon. For example, Newman et al. (2005), found bacterial community decomposition on leaves was affected by sedimentation and that anaerobic species dominated at sediment impacted sites. Whereas, Dickman et al. (2005) found that sedimentation has little effect on motile unicellular diatoms and filamentous cyanobacteria which move above the sediment layer.

Although motile unicellular diatoms and filamentous cyanobacteria contribute to primary production, they are less palatable to invertebrates; potentially altering energy pathways. Moreover, deposited sediment can become fused into biofilms (i.e., algae, bacteria and fungi). Sediment incorporated within the matrix decreases *chlorophyll-a* concentrations, this in turn reduces photosynthetic ability. Again, this potentially alters energy pathways by reducing the nutritional value to invertebrates. Furthermore, sediment also acts as a major carrier of other pollutants, including phosphorus and toxic metals (Graham 1990). As sediment becomes fused into the biofilm, the associated toxins increases resulting in further changes to resource pathways (Muirhead et al. 2004). Graham (1990), showed that even at low concentrations sediment containing toxins accumulates in periphyton.

Fine sediment has low inertia, therefore can be easily re-entrained by increasing flows. Once particles begin to move the substrate becomes unstable. Substrate instability has a negative effect on microbial communities which are dislodged. A 2003, study in two West Coast stream, one which associated with gold mining, showed that periphyton communities were mostly restricted to stable substrates such as cobbles in riffles, and were almost absent in runs that were dominated by loose shifting sands.

Fate of the primary consumers (e.g., macro-invertebrates, snails and others)

Deposition can directly bury or reduce the food quality for invertebrates (Luedtke and Brusven 1976, Richards and Bacon 1994, Danger et al. 2012). One of the earliest noted effects of increased deposited sediment was in 1936 when increased mortality in aquatic bivalves was noted as a consequence of rapid burial (Ellis 1936). Burial can have a direct effect on sedentary animals especially where rates of deposition are high. Communities within the hyporheic zone may be particularly susceptible to smothering due to infilling of interstitial spaces. A consequence of burial is decreased oxygen flow (Richards and Bacon 1994) at both the bed and in the hyporheic zone. Furthermore, burial can also reduce food quality, detrital matter (e.g., leaf litter) is less palatable to shredding invertebrates when buried or smothered. In Emerald Creek, North Idaho, Luedtke & Brusven (1976) showed that sand deposition prevented the upstream movement of invertebrates, because the substrate lacked stability. After flood disturbance, invertebrate densities and diversity were significantly higher on stable as opposed to unstable substrates. (Matthaei et al. 2000).

Since the mid 1900's the impacts of sediment on aquatic macro-invertebrate communities had been widely reported, potentially because invertebrates especially Ephemeroptera, Plecoptera and Trichoptera (known as EPT) were used extensively to assess pollution in freshwaters. The ubiquity, high diversity and range of sensitivities of macro-invertebrates made them the most practical group for monitoring waterway health.

EPT are typically negatively correlated with increases in fine sediment. Sediment driven changes are often characterised as a change from assemblages of EPT to communities dominated by snails, chironomids, and oligochaetes (Waters 1995). For example, in the Appalachian streams of North America, Angradi (1999), showed that families such as the larval Baetidae and Orthocladiinae, had positive reactions to sediment increases, becoming proportionally more abundant within the community. In the same study other benthic invertebrate densities were reduced by a factor of 2 when the percentage of fine sediment bed cover increased from 0 to 30% (Angradi 1999). Research has shown non-sensitive taxa have low oxygen requirements and are often deposit feeders whereas, sensitive taxa are often filter-feeders and scrapers with higher oxygen requirements (Hynes 1970, Lemly 1982, Waters 1995).

The New Zealand caddisfly (*Pycnocentrodes spp.* McLachlan) and the mayfly (*Deleatidium spp*. Eaton), prefers un-silted periphyton, displaying reduced growth rates when fed periphyton containing sediment (Ryan 1991). Experiments conducted by Broekhuizen, Parkyn and Millar (2011), concluded that intermediate levels of sediment contamination within food resources supported increased growth in the New Zealand mud snail (*Potamopyrgus antipodarum,* Gray). This potentially allows the mud snail to dominate communities at sedimented sites (Broekhuizen et al. 2001). Mud snails are highly defended by a shell and operculum which fully shields the aperture and protects the snail from dissection (Winterbourn 1970). The shell and operculum allows mud snails to survive passage through the gut of fish (Bruce 2006). Their defences and propensity to consume and therefore contain sediment makes them a low quality food source for fish.

In 2013, an intensive study of 30 Canterbury agricultural waterways concluded that pollution sensitive taxa (Ephemeroptera, Plecoptera, Trichoptera), were significantly reduced at bed sediment cover above 20% (Burdon et al. 2013) Modelling of variables describing sediment affects showed habitat loss was the key driver of changes to invertebrate communities (Burdon et al. 2013). This was linked to the quality of the riparian margin which controlled sediment inputs and therefore instream habitat. Reductions in the quality and availability of basal resources has potential flow on affects, including reductions in the density of macro-invertebrate grazers and scrappers.

Although they may be highly abundant in sedimented waterways, non-sensitive taxa show a low propensity to drift and are often highly defended (e.g., mud snail) burrowing taxa (e.g., oligochaetes) (Burdon et al., 2013). Therefore they are less likely to provide an equivalent food resource for fish. An increase in the abundance of aquatic invertebrates that are less vulnerable or attractive to fish, should result in altered predation pressure and/or reduced fish abundance or condition. Although a range of responses to high deposited sediment have been noted in New Zealand invertebrate communities, little is known about the responses of native fish assemblages to sedimentation (Burdon et al. 2013).

Apex predators and deposited sediment

Sediment filling interstitial spaces increases habitat homogeneity (Fig. 1.4) by reducing the distinction between riffle, run, and pool geomorphology. As a consequence there is less refugia for fish and crustaceans (Yarnell et al. 2006). A lack of refugia was shown to increase the energy requirements of prey species or lead to reduced survival from predation (Franken et al. 2006).

High sedimentation reduced brown trout spawning sites (Scheurer et al. 2009). Brown trout rely on benthic substrate (Wood and Armitage 1997) with a percentage of intra-gravel sediment that does not exceed 10%, above this threshold salmonid egg survival can rapidly decrease (Jensen et al. 2009). Alternatively, smothering of the redd, can adversely affect newly hatched fry by lowering the dissolved oxygen (Cottam and James 2003). Furthermore, the occurrence and density of trout, especially fry (less than one year), was affected by the presence of medium-sized substrate (Eklov 1999). In some instances sedimentation leads to increases in parasite-transmitting vectors. Brown trout populations decreased due to the spread of the lethal whirling disease *M. cerebralis* carried by the *T. tubifex*, which is abundant in highly sedimented sites (Brinkhurst 1996).

International literature suggests, insectivores fish growth rates, survival, distributions, biomass, abundance and diversity decrease as the composition of the benthic invertebrate fauna changed with sedimentation (Alexander and Hansen 1986, Berkman and Rabeni 1987, Osmundson et al. 2002). A similar affect was seen in the abundance of the blue ducks; *(Hymenolaimus malacorhynchos,* Gmelin) who share the role of top predator in New Zealand waterways with fish. A volcanic eruption in the North Island, deposited a significant load of sediment into nearby streams. The increased sediment resulted in reductions in invertebrate quantity and quality. Blue ducks, who fed on the aquatic invertebrates, were directly affected by reductions in invertebrate prey (Collier 2004).

A comparison of two West Coast sites, one impacted by gold mining elevated sediment inputs, the other without, indicated that increased turbidity and deposited sediment, resulting in reduced food resources, were very likely to have affected fish community composition. Furthermore, it appears there was a link between sediment cover and fish density, with lower densities in high sediment sites (Cottam and James 2003).

New Zealand freshwater crayfish appear to be affected by deposited sediment, although they are known to be active burrowers (Whitmore and Huryn 1999), sometimes hiding in the crevices in banks. In an example from the sediment literature, a study of 30 pools sited along a stream in Otago, the abundance of juvenile Southern crayfish was associated with coarse substrate (Usio and Townsend 2000). The abundance and distribution of adult crayfish is also known to controlled by substrate (Capelli and Magnuson 1983). Moreover, New Zealand crayfish occupy the trophic position of a top predator, by functioning as omnivores, they fill dual roles as predators and processors of organic matter (Parkyn et al. 2001). The crayfish's omnivorous nature and its use of habitat that could be infilled by sediment makes it a potential sediment excesses indicator taxa.

The majority of literature described the negative impacts of deposited sediment on apex predators (such as fish and crayfish) to both fish communities composition and individual taxa therefore, I hypothesis that deposited fine inorganic sediment would cause reductions in New Zealand fish diversity and abundance by a combination of both bottom-up affects, driven by changes to the invertebrate community and the direct effects of habitat heterogeneity should also drive fish assemblages. Habitat changes are particularly important to New Zealand's fish fauna as the majority are benthic.

TURBIDITY - THE IMPACTS ACROSS MULTIPLE TROPHIC LEVELS

Some human activities result in prolonged turbidity, (e.g., mining, gravel quarrying) (Parkhill and Gulliver 2002), or increase the frequency of pulse events, (e.g., mechanical clearance of macrophytes)

(Greer et al. 2016). Stream biota have evolved to withstand natural pulses and background levels of suspended sediment. However, once the norms are exceeded, turbidity becomes a biotic stressor, affecting population performance, community composition and ecosystem functions.

The effects of turbidity on basal resources

High suspended sediment and the resulting elevated turbidities are known to reduce primary production by effecting photosynthesis (Ryan 1991) and by abrading algal foliage (Wood and Armitage 1997, Luce et al. 2013). When transported in as bedload via saltation, fine sediment damages or removes aquatic plants and algae through abrasion. For example using *in situ* experiments Luce et al. (2013) measured reductions in periphyton biomass caused by increases in sediment transport rates.

Increasing turbidity attenuates light, reducing the photic zone (Kristensen and Suraswadi 2002) (Fig. 1.5). The relationship between turbidity and light attenuation indicates that a turbidity of 250 NTU will attenuate about 95% of light in a water column of 0.5 m depth (Stevenson et al. 2010). Turbidity scatters light reducing light penetration. Prolonged reduction in the amount of light that can penetrate the water column can reduce primary production in streams. Ryan (1991) investigated the impact of turbidity on in-stream productivity and found that low turbidity, of only 5 NTU, decreased primary productivity by 3-13%.

In New Zealand reduced light through-out the water column has been shown to affect photosynthesis and plant growth (Davies-Colley et al. 1992). Reductions in algal biomass and overall primary productivity have flow-on affects to higher trophic levels. Reductions in primary productivity and the associated decreases in algal biomass, phototrophic content (Davies-Colley et al. 1992) and possibly changes in epilithon assemblage (Newcombe and Macdonald 1991) can have flow on effects on higher trophic levels, such as grazing invertebrates.

The direct effect of turbidity (suspended sediment) on aquatic macro-invertebrates

Increased turbidity has been described as the strongest descriptor of reduced invertebrate density and biomass and has been linked to changes in community assemblage. High turbidity reduces invertebrate densities by clogging feeding and respiratory organs, directly effecting the predation capacity of visual predatory invertebrates and changing the behaviour of aquatic invertebrates (Arruda 1983, Culp et al. 1986).

Behavioural changes include increased drift rates, apparently without an increase in upstream movements. In Ryan's 1991 sediment review, he referred to Rosenberg & Wiens (1975), who were the first to demonstrate that increased suspended sediment levels were associated with increased drift rates in Canada. Interestingly, some researches considered the active component of invertebrate drift was actually an attempt by invertebrates dislodged, by turbidity, to swim back down to the substrate (LaPerriere 1983). However, instead of reattaching to the substrate, the increase in turbidity may cause the drift fauna to continue to drift, thus reducing invertebrate biomass. Normally, downstream drift would be compensating by movement upstream, when this is not the case biomass is decreased. However, reductions may also lead to altered taxonomic composition. Field experiments conducted by Culp et al. (1986), showed that suspended and deposited sediment increased invertebrate drift to “catastrophic levels”. However, drift rates were variable among invertebrate species, leading to changes in community assemblage. For example, mayflies are known to have a high propensity to drift reducing prey availability. In streams affected by alluvial gold mining on the West Coast of New Zealand Quinn et al. (1992) found that high turbidity decreased invertebrate densities.

Large sediment particles suspended during high flows can damage unprotected fleshy body parts, such as gills, making respiration difficult. Filter-feeders are particularly prone to damage by suspended particles which clog feeding apparatus, (e.g., the mesh of net-spinning species). To avoid injury (and possibly because of reduced food quality) many invertebrates respond to high turbidity by finding refuge or withdrawing sensitive structures such as filter combs. For example, both freshwater molluscs and Cladocera eject unwanted sediment particles from feeding organs, spending an increasing amount of time on this cleaning behaviour as turbidity increases (Arruda 1983). Once structures (gills or feeding organs) are clogged they must be cleaned or feeding or respiration will become reduced.

Effects of the turbidity on fish

High concentrations of entrained sediment result in gill abrasion (Herbert and Merkens 1961) and reduction in oxygen. At the extreme end of the scale high concentrations of suspended sediment can prove lethal to fish (Allan et al. 2007, Kemp et al. 2011). For example the median lethal turbidity level (LC50) over 24 hours for īnanga was determined as 20,235 NTU, but for smelt it was as low as 3,050 NTU (Hicks et al. 2009).

Sediment contains high concentrations of organic matter which has a high affinity for oxygen. Excessive sediment can lead to deoxygenation of the water column (Bruton 1985) resulting in fish mortality (Kemp et al. 2011). To date, there has only been one documented occurrence in New Zealand, when high sediment loads from silviculture, killed numerous fish in Marlborough (Allan et al. 2007). However, these events will increase in frequency as sediment inputs and legacies evolve, driven by further land-use intensification and climate driven flow reductions.

Despite this, the majority of affects attributed to turbidities are non-lethal; and a reduction in water clarity due to increased turbidity can affect feeding and migratory behaviour (Boubée et al. 1997, Dean and Rowe 1998).

Elevated turbidity leads to avoidance by fishes, even in the presence of suitable habitat and food resources (Boubée et al. 1997). Avoidance potentially reduces recruitment, affecting the whitebait fishery. More than half of New Zealand fishes are diadromous, selecting a waterway upon return from the ocean. River selection by migratory juveniles is known to rely on pheromonal cues from adults (Baker and Hicks 2003). Nevertheless, low turbidity water (<25 NTU) has been shown to lead to avoidance by some taxa (Boubée et al. 1997).

The rainbow trout were found to be sensitive to turbidity, reactive distance decreased to zero linearly between 5 and 50 NTU (Barrett et al. 1992). Conversely, Rowe et al. (2003) undertook experiments that found that turbidity had no effect on the feeding rate for rainbow trout.

Many New Zealand fishes are usually considered visual predators, feeding on prey drifting in the water column. However, Dean and Rowe (1998) found that banded kōkopu and common bullies have a comparatively well-developed ability to feed using non-visual senses. As New Zealand fish are for the most part nocturnal hunters, it is logical that they rely heavily on non-visual senses. Increased turbidity amplifies invertebrate drift rates this should provide increased feeding opportunities for fish. Nevertheless, the high turbidity simultaneously reduces visibility which impairs visual feeding, negating any benefits for fish, unless species possess non-visual senses. Williams et al. (2002) conducted tank experiments which suggested that the feeding (*Daphnia spp.)* rate of adult smelt and īnanga were not affected at 160 NTU (Williams et al. 2002). In 2008 testing īnanga and kōaro found that the growth of īnanga was effected by turbidity between 5 and 15 NTU after 21 days. Similarly, kōaro growth was affected at turbidity between 15 and 50 NTU. There was no apparent effect on weight of these species with turbidity levels up to 200 NTU

They showed highly variable results at the various different turbidity levels. However, it is likely that very high turbidity (e.g., > 4000 NTU), severely disrupts feeding capacity. Reduced feeding rates resulted in reduced growth rates and negatively impacted fish condition in trout (Cavanagh et al. 2014).

The response of banded kōkopu to suspended sediment appears to differ between migratory juveniles and adults (Boubée et al. 1997). The same variable tolerances to turbidity at different life stage may occur in other fishes. The turbidity responses of several juvenile native fishes have been investigated. Unfortunately little research has been undertaken into the response of adult fish to sediment, necessitating further research into threshold responses to turbidity of New Zealand freshwater fishes. Determining the extent of deleterious impacts on fishes feeding is important for conversation and can further our understanding of the impacts caused by anthropogenic activity.

SCALAR AFFECTS

Studies worldwide, have related various habitat factors to communities by using measures of diversity or abundance. The factors that determine fish distributions and abundances in New Zealand vary with scale, making it an important factor to consider when seeking to determine broader community thresholds to sediment. At small scales, such as the reach, the physical factors that determine community assembles include habitat diversity and size, and distance from the sea. However, this changes at the regional or catchment scale, where diversity is attributed to catchment size and river discharge. At the largest scales, (e.g., continental), diversity is attributed to climatic events or the geography (Lévêque et al. 2005). Despite studies indicating strong scalar affects being common, a patch scale survey would not provide insight into fish diversity or abundance. Conversely, the catchment scale is too large for monitoring by local bodies and likely to hide any sediment driven affects.

Community effects

Non-random loses of taxa, as a result of both deposited and suspended sediment, can lead to less specious communities (Fig 1.6). Changes to the dominant invertebrate taxa and reduced species richness, generates simpler food webs (Pimm and Lawton 1977), reducing and altering resource availability and quality, for fish (Sagar and Glova 1998) and distorting interaction strengths (Tscharntke et al. 2007). These changes make communities highly unstable (Tilman et al. 1997) and potentially modify the distribution and abundance of fish.

A fundamental characteristic of food webs is ‘food-chain length’ which refers to the number of transfers of energy or nutrients from basal groups to the top predators. Sediment in both forms (suspended and deposited), may lead to shorter food chain length through reduced primary productivity or other basal resources, decreased invertebrate biomass or diversity can change and reduce the abundance of prey (quantity or quality) for apex predators. Perturbations that affect one trophic level of the food can affect biota at higher trophic levels (Petchey et al. 2004). Moreover, sediment, deposited and suspended, affects all trophic levels from primary productivity to apex predators.

In New Zealand naturally short food chain lengths (Harding et al. 2004) have elevate fish and crayfish to the status of dominant predators (McDowall 2000), making them critical for food web stability and suggesting any negative affect of fine inorganic sediment may be magnified at that trophic level (Thompson and Townsend 2005).

SEDIMENT DISTURBANCE

Disturbance strongly shapes community composition (Resh et al. 1988). Resh et al. (1988) in their review concluded: ''*In fact, to some of us, disturbance is not only the most important feature of streams to be studied, it is the dominant organizing factor in stream ecology*''. Sediment (in both forms entrained and deposited) is linked to water velocity and therefore to ecosystem disturbance.

Disturbances can differ in their temporal pattern, intensity and duration, referred to as pulse, press and ramp. Pulses disturbances are of short duration and intense, such as major floods in constrained river channels which bring a pulse of turbidity. They are often followed by pulse responses for example the response to a floods high turbidity pulse may be an initial rapid reduction in algal biomass followed by a rapid return to pre-disturbance state.

Press disturbances rise sharply and then reach a constant level that is maintained over long periods. This occurs with many human-induced disturbances, for example, suspended sediment concentrations increased by 15,687 mg L^{-1} immediately after mechanical macrophyte clearance. The removal of the macropytes then saw significant long-term, (77 days) increases in suspended sediment. Without the macrophytes, exposed bed material was continually re-suspended even in moderate flows (Greer et al. 2016). Furthermore, if the macrophyte clearance process is repeated on a regular basis (e.g., the middle and end of the growing season), the disturbance may produce a ramp effect, a situation where disturbance strength steadily increases over time. Ramp disturbance may also occur with sediment inputs from by non-point sources such as tile drain and wastewater. Continued sediment ramping will lead to a legacy of deposited sediment.

The importance of disturbance studies in benthic ecology comes from trying to understand the communities response i.e., press disturbances often instigate press responses in biotic communities and likewise ramp disturbance can produce ramp response. However, mirrored responses are not always the case, a pulse disturbance may produce a press response and a press disturbance may produce a ramp response. Despite responses being varied disturbance directs community composition in New Zealand freshwater systems. Sampling of 54 New Zealand streams indicated bed stability was a strong predictor of community composition, with a very specific flow disturbed community being identified.

Disturbance can be linked to community assemblage with intermediate disturbance hypothesis (IDH), which predicts moderate amounts of disturbance provide the widest variety of habitats and therefore has the highest diversity. If this theory is correct intermediate levels of deposited sediment, increase habitat heterogeneity by providing a greater number of niches and should therefore, increase fish diversity.

The dynamic constraints hypothesis suggests disturbance disproportionately affects higher trophic levels. Because species higher in the food chain are rarer, they are more likely to be lost from the system during a disturbance event (Thompson and Townsend 2005). Therefore, there is likely to be a negative effect on native fish community assemblages driven by increased deposited sediment. If dynamic constraints hypothesis (Thompson and Townsend 2005) is underlying community response to increasing sediment, we should expect to see notable changes in fish assemblages (diversity and abundance), as fish represent the highest trophic level in aquatic systems in New Zealand; with species populations being lost from highly degraded sites.

The feeding rates of New Zealand freshwater fishes were expected to be negatively affected by turbidity. I expected increasing turbidity would reduce feeding success rates, by impairing the visual cues fish use to detect and capture prey and that different taxa would show varied tolerances.

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CHAPTER 2

THE STRUCTURE OF NEW ZEALAND BENTHIC FISH ASSEMBLAGES IS LINKED TO DEPOSITED INORGANIC SEDIMENT

Homogenisation of habitat by deposited sediment reduces refugia and can lead to increased risk of predation. Despite its cryptic colouration this member of the family Eleotridae is quite obvious on the highly sedimented stream bed.

ABSTRACT

Freshwater ecosystems depend on a dynamic flux of resources and organisms from the land and sea. However, this provides an opportunity for negative subsidies such as pollutants, including excessive fine inorganic sediments, to enter the system. The harmful impacts of excessive anthropogenically introduced sediment on freshwater ecosystems can be significant. Deposited sediment is known to disrupt aquatic resource pathways and homogenise habitats. Despite the disruptive nature of sediment, few New Zealand studies have determined the impacts on diversity (species richness) and abundance (density of fish) freshwater fish assemblages.

I examined the effects of fine inorganic bed sediment on benthic fish communities in 52 waterways in the South Island of New Zealand. Community richness was negatively related to bed sediment cover. Waterways with low bed sediment cover had up to eight fish species, whereas highly sedimented waterway (i.e., > 80%) had a maximum of four species. One waterway which had 100% bed sediment cover had no fish present. The relationship between sediment cover and species richness was non-linear suggesting the fish community may display a threshold, at which point the community assemblage changes from sediment-sensitive to sediment-tolerant species.

Multi-dimensional modelling of the data established sediment was only one of a multitude of factors, such as disturbance, pH, shade, discharge, salinity and specific conductivity, which play important roles in determining fish assemblages. Furthermore, the model gave an indication of which species may be more sediment-tolerant (e.g., shortfin eels (*Anguilla australis,* Richardson) and elvers).

Structural modelling highlighted the mechanisms by which deposited sediment reduces diversity and abundance (i.e., habitat homogenisation, reductions in sediment-sensitive invertebrates and discharge). The model showed complex interactions between shade (an indicator of the quality of the riparian margin), sediment increases and both fish and sensitive invertebrate taxa declines. These results suggest that activities that increase sediment deposits in waterways will have a negative impact on native fish communities in New Zealand.

INTRODUCTION

Freshwater systems receive fine inorganic sediment from natural processes, including in-stream abrasion, erosion and aeolian (wind derived) sources. Variations in natural sediment supply are influenced by catchment geology, topography, vegetation type and cover, rainfall, and catastrophic events, such as volcanic eruptions, earthquakes or extreme storms (Hicks and Griffiths 1992). However, surveys conducted by the U.S. Environmental Protection Agency emphasised that increased sedimentation, related to anthropogenic activity (such as land-use change), was one of the most pervasive and important types of river and stream pollution (Waters 1995).

Worldwide, most waterways are now subject to high sediment loads on both a continual and episodic basis (Hicks and Griffiths 1992). The multifaceted degradation of streams draining urban land (including by sedimentation), has been described as 'urban stream syndrome' (Walsh et al. 2005). While expansion and intensification of agriculture and horticulture have led to further sediment inputs. Over decades, altered flow regimes combined with increased sediment inputs (Walling 1983) have resulted in a build-up of deposited sediment (Fryirs 2013), generating a sediment legacy. Once fine inorganic sediment accumulates on the bed, its effects on biotic communities can range from minor to cataclysmic.

Sedimentation of a waterway can change and reduce physical habitat for benthic macro-invertebrates and fishes. Sediment filling interstitial spaces increases habitat homogeneity by reducing the distinction between riffle, run, and pool geomorphology. Yarnell et al. (2006) used flume and field studies to evaluate the relationship between sediment inputs and habitat heterogeneity in the South Yaba River, California. They determined that sediment filling interstitial spaces increased habitat homogeneity and reduced hydraulic roughness. Homogenised habitats provided less refugia from predation for both fish and invertebrates (Yarnell et al. 2006). Additionally, a lack of refugia was thought to increase the energy requirements of prey species or lead to reductions in survival rates from predation (Franken et al. 2006). Likewise, sediment induced decreases in hydraulic roughness will increase water velocity, increasing the requirement for refuge from the intensified current (Rempel et al. 2000).

Sedimentation not only reduces habitat heterogeneity it simultaneously disrupts resource pathways, by impairing both primary production (Dickman et al. 2005) and detrital breakdown (Danger et al. 2012). Furthermore, sediment negatively impacts the biomass of algae by reducing algal habitat and smothering algal communities (Wood and Armitage 1997).

Once interstitial spaces are filled, sediment accumulates, disconnecting the hyporheic zone from the water column (Nogaro et al. 2010). This disconnection reduces the amount of dissolved oxygen in the hyporheic zone and creates low oxygen or anoxic conditions. Anoxic and hypoxic conditions have been shown to alter microbial activity and change the taxonomic composition of the epilithon (Nocker et al. 2007).

Deposited sediment often forms complex mattresses that can become fused into biofilms. This reduces the nutritional value while simultaneously increasing toxins and faecal contamination of the biofilm (Muirhead et al. 2004). By conducting feeding trails in a French second order stream, Danger et al. (2012), examined the effects of a range of burial conditions on decaying leaf litter palatability to invertebrates. They determined that detrital matter (e.g., leaf litter) was less palatable to shredding invertebrates when buried or smothered. Reductions in the quality and availability of basal resources have flow on affects, including reductions in the density of invertebrate grazers and scrappers (Broekhuizen et al. 2001).

In New Zealand, agricultural intensification, urbanisation, and extractive industries, such as forestry and mining, are major sources of sediment in waterways (Ballantine and Davies-Colley 2014). Increases in sedimentation have been shown to affect New Zealand biota at a range of trophic levels. For example, the New Zealand caddisfly (*Pycnocentrodes spp.* McLachlan), displayed reduced growth rates when fed periphyton containing sediment (Ryan 1991).

The diversity and structure of invertebrate communities has been shown to be a good indicator of waterway health (Stark 1998). Invertebrates are impacted by increased sediment which reduces sensitive taxa (e.g., Ephemeroptera, Plectoptera, Trichoptera) (Burdon et al. 2013). Certain biotic traits, allow dominance by a few highly defended or burrowing, sediment tolerant generalists (e.g., snails, nematodes, chironomidae larvae, oligochaetes and bivalves) (Broekhuizen et al. 2001, Burdon et al. 2013). Experiments conducted by Broekhuizen et al. (2011), concluded that intermediate levels of sediment contamination within food resources supported increased growth in *Potamopyrgus antipodarum,* Gray. This allowed *P. antipodarum* to dominate assemblages at highly sedimented sites (Broekhuizen et al. 2001, Burdon et al. 2013). Although they may be highly abundant in sedimented waterways, non-sensitive invertebrate taxa, such as *Potamopyrgus*, show a low propensity to drift (Burdon et al., 2013). Therefore, they are less likely to provide an equivalent food resource for higher trophic levels.

Although a range of responses to high deposited sediment have been noted in New Zealand invertebrate communities (Collier et al. 1998, Broekhuizen et al. 2001, Death et al. 2003, Hicks et al. 2004, Burdon et al. 2013), little is known about the responses of native fish assemblages to sedimentation. However, coarse bed substrate and interstitial refugia are known to be critical for some New Zealand native fishes because the majority are benthic, using the streambed for refuge, foraging, and in some cases nesting.

A single study restricted to the Waikakahi Stream in South Canterbury indicated that fish distributions were not correlated with reach-scale riparian or instream habitat variables, such as sediment. However, this study was undertaken in waterways that had historically been impacted by high sediment loads from agriculture. The study sort to identify any positive changes associated with improved riparian management to brown trout (*Salmo trutta,* Linnaeus) and eel (Anguillidae) populations. However, fish densities were potentially too low to detect any effect of sediment reductions (Holmes et al. 2016) and the legacy of dysfunction attributed to agricultural use will likely be long lasting (Harding et al. 1998). However, in mine impacted waterways on the West Coast reductions in fish diversity were directly linked to increased fine sediment (Cottam and James 2003).

International literature suggests insectivores fish biomass, abundance and diversity decreased as the composition of the benthic invertebrate fauna changed with sedimentation (Alexander and Hansen 1986, Berkman and Rabeni 1987). In contrast there is limited data available on which New Zealand fish taxa are tolerant to the effects of deposited sediment, or whether New Zealand native fish populations are more adversely affected by sedimentation than introduced sport species, such as Salmonids.

In this chapter, I examined the effect of increasing sedimentation on the fish communities in waterways throughout the South Island of New Zealand. The aim was to determine if high levels of sedimentation would alter fish community assemblages (i.e., diversity and abundance). Studies have already shown, highly sedimented waterways contain very different invertebrate communities to low sediment waterways. Therefore, I hypothesised that increased deposited sediment cover would decrease the abundance and/or diversity of freshwater fish species, resulting in altered community composition. Furthermore, I expected that excessive sediment might impact fish communities by both direct and indirect pathways, such as altered prey (quality, or quantity) and habitat homogenisation (Fig. 2.1).

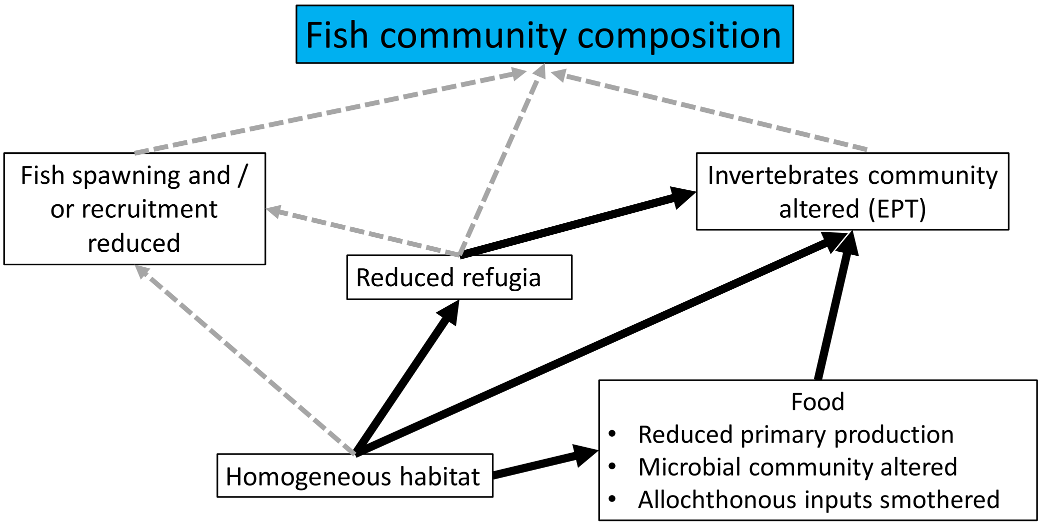


Figure 2.1. Conceptual diagram illustrating known and hypothsised links between sediment and fish community. The black arrows represent known links between deposited sediment and altered invertebrate assembalges (Burdon et al. 2013). The broken grey arrows represent hypotheised links, both direct and indirect between deposited sediment and fish assembales.

METHODS

Selecting study sites

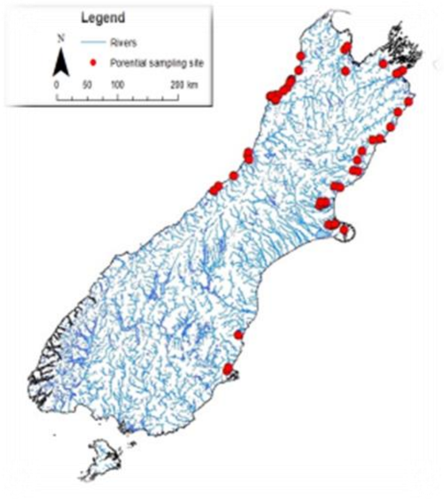


Figure 2.2 Map of the 52 sites sampled throughout the South Island of New Zealand.

A total of 187 potential sites throughout the South Island of New Zealand were identified using Google maps. The waterways finally sampled were selected along the following criteria: first to third order (wetted width 0.5 - 6 m), perennially wet, all were wadable (depth < 0.5 m) and had < 40% macrophyte cover to allow affective fish sampling. So from the 187 sites, a total of 52 waterways were selected to be sampled (Fig. 2.2). To provide greater inference my survey area included coastal sites in the upper South Island (site locations are summarised in Appendix D). Unfortunately, the occurrence of an El Niño Southern Oscillation coincided with the sampling period, the austral spring summer (October 2015 to February 2016). This weather pattern dried streams along the south eastern edge of the South Island, from Timaru to Invercargill. Therefore, fewer waterways on the east coast fitted the sampling criteria.

Studies have illustrated the importance of elevation and distance from the ocean in controlling New Zealand fish distribution. In New Zealand fish diversity is greatest closer to the sea, due to diadromy. To ensure any links between sediment and diversity are not compromised by diadromy, all reaches sampled were situated close to the coastline (<2 km). This nearness to the sea also increases the likelihood of unobstructed ocean access for fish migration. The New Zealand fauna with its high proportion of diadromous species has an enormous potential to be affected by barriers to migration. Furthermore, studies have also illustrated the importance of elevation in controlling New Zealand fish distribution, so waterways were located at elevations <220 m (Strickland et al. 2001).

Waterways surveyed represented a gradient of sediment bed cover from <5% to 100% (Appendix E). To gain an orthogonal data distribution across the sediment gradient I initially determined each waterway suitability from the bankside using a deposited sediment-cover visual assessment along the 20m reach (Clapcott et al. 2011).

In each waterway a 20 m reach was selected which included a run–riffle sequence with pools, when present. Galaxiidae and eels (Anguillidae) often occur in deep water habitats, so where possible these were included (Chadderton and Allibone 2000).

Waterways selected were sampled on a single occasion under base flow conditions. I avoided sampling reaches near confluences or reaches affected by structures (e.g., bridges, weirs or access points for gravel or water extraction) as these potentially generate disturbance which is known to directly impact fish communities (Jellyman et al. 2013).

Once sites were selected catchment type was indicated as national park, dry-stock, dairy, viticulture, or forestry. Sites were noted as being on the east or west coast of the South Island as the Alpine Divide, a mountainous region running the length of the island, alters climatic conditions (e.g., rainfall and temperature).

Measuring sediments

Sediment cover in each reach was assessed by three methods. Firstly a visual bankside assessment of percentage bed cover for the whole reach was estimated. Sediment assessment protocols suggest that visual bankside estimates of percentage sediment bed cover are strongly correlated to physical measures of deposited sediment (Clapcott et al. 2011). Secondly, estimates of sediment cover were made by visually assessing five haphazardly placed quadrates (30 cm x 30 cm), with the aid of a bathyscope. Lastly, the intermediate axis of 50 random substrate particles was measured using a gravelometer. Particles were selected by walking in a zig-zag pattern up the 20 m reach referred to as a 'Wolman walk' (Wolman 1954).

Sediment depth measurements were taken centrally within the five quadrates. Sediment depth was assessed in each quadrat by inserting a graduated instrument into the sediment until the underlying, coarser substrate was reached.

Fish & crayfish sampling

Reaches were sectioned off with a single, 2 mm mesh stop net at the top and bottom to create a closed system prior to sampling.

Fish and crayfish were sampled with a portable (KAINGA EFM300) electric fishing machine by completing three passes of a 20 linear meter reach. Captured fish were identified to species level where possible, counted and their size noted in the field. Very small fry (> 4 cm) were identified to family. Glass eels and elvers (Anguillidae) (>10 cm) were recorded as elvers.

Invertebrate collection

A single invertebrate kick net sample was collected from five representative habitats within the reach using kick-net 500 µη mesh. Samples were labelled and stored in 70% ethanol. In the laboratory the samples were passed through a 500 µη Endecott sieve. All invertebrates were removed, identified to the lowest practicable level (usually genus) using invertebrate identification guides (Winterbourn

2006). Relative abundances of taxa on a "rare", "common", "abundant", "very abundant", "very, very abundant" scale (i.e., R, C, A, VA, VVA), were recorded.

Physio-chemical measurements

At each waterway, spot water chemistry was recorded, including pH, specific conductivity, dissolved oxygen (i.e., % and saturation), and temperature using standard meters: YSI63 and YSI550.

Discharge was calculated from a single transect which included three measurements of wetted width, depth and velocity which was measured with a Marsh-McBirney Flo-mate (Model 2000; Hach, Loveland, Colorado, USA)

A channel stability assessment Pfankuch (Pfankuch 1975, Collier 1992) was conducted, providing estimates of bank stability, bed stability and flood disturbance. This involved scoring 15 variables (weighted in relation to their perceived importance). The scores were combined to give an overall stream stability score. Percentage cover estimates of macrophytes (bed and bank), algae, moss, woody debris and leaf packs were made using five haphazardly placed quadrates (30 cm x 30 cm). Shading was measured using a spherical densitometer above each quadrat and bank undercuts (cm) were measured on each side of the quadrat.

Data analysis - general

The three sediment measures, (i.e., visual bankside assessment reach percentage cover, quadrates and the Wolman walk), were averaged to create a substrate index. Shade, undercut, macrophytes (bed and bank), algae, moss, woody debris and leaf packs, which were assessed within the five quadrates, were averaged.

Invertebrates were assessed using the semi-quantitative macroinvertebrate community index (SQMCI) (Stark 1998) and EPT taxa counts (Appendix F). From quantitative electrofishing results, I estimated the abundance of each fish taxa by maximum weighted likelihood with the Carle-Strub depletion estimator (Carle 1978.) (Appendix G).

Relations between deposited sediment and fish assemblages

Statistical analysis was undertaken using R statistical software (Team 2014). All statistics were considered statistically significant where $P < 0.05$. Initially data sets were $\log_{10}$ transformed, or in the case of percentages logit transformed, to ensure normality and homogeneity of variance for statistical analysis. This was followed by exploratory analyses undertaken by constructing linear regressions (with linear, quantile and quasi-poisson terms).

Traditional regression analysis was used to determine any significant relationships between sediment depth or bed sediment cover and fish density and species richness, at the 52 sites (including the one where fish were absent). Analyses showing data over-dispersion were re-analysed using quasi-poisson regression. Due to the complexity of the interactions variances were not heterogeneous, implying there was not one single rate of change, therefore quantile regression (Koenker 1978) was used to more accurately portray the limiting effects of sedimentation on species richness.

Variables driving fish community assemblage

Fish assemble data and environmental variables were moved into multidimensional space and ordinated by non-metric multidimensional scaling (NMDS). The ordination was performed in R (Team 2010) using the envfit function, vegan package and monoMDS (Oksanen 2015). One site, Birchwood, was omitted from the ordination analysis because no fish were present inside the study reach.

Smelt (*Retropinna retropinna*, Richardson), yellow-eyed mullet (*Aldrichetta forsteri,* Valenciennes) and īnanga (*Galaxias maculatus,* Jenyns) are pelagic species (Jellyman 2010), and therefore their interaction with sediment varies from that of benthic species. The affective use of freshwater fishes as bio-indicators requires that the taxa used is resident or sufficiently exposed to site conditions to elicit a biological response. This suggests pelagic species (smelt, yellow-eyed mullet and īnanga) lack suitability for this analysis. Consequently, these species were not included in the analysis of effects of deposited sediments on community structure. Species with extremely low abundances (e.g., black flounder *(Rhombosolea retiaria,* Hutton*)* and torrentfish *(Cheimarrichthys fosteri*, Haast*)* were also not included.

The NMDS used rank orders Bray-Curtis dissimilarity matrix (Quinn and Keough 2002) to examine the relationship between the fish assemblage at the 51 sites were fish were present and the physical variables including percentage deposited sediment, salinity, pH, shade, specific conductivity, discharge and disturbance (using physical stability assessed by the method of D.J. Pfankuch).

Modelling structural drivers of fish assemblages

Structural equation modelling (SEM) (Fox 2010) was used to estimate direct and indirect affects between selected physical and biotic variables in causal path models (Grace 2006). Specifically, I used SEM to investigate potential mechanistic pathways influencing fish assemblages as mediated by deposited sediment. The analysis was performed in R (Team 2014) using the functions, ape - Version 3.3, caper - Version 0.5.2, nlme - Version 3.1.122, lavaan - Version 0.5.19, piecewiseSEM - Version 1.0.0, vegan and car.

Tests for normality were conducted using Shapiro–Wilk test (Trifković 2015). Data was log-$\log_{10}$ or logit transformed (Warton and Hui 2011) to ensure normality and homogeneity of variance for statistical analysis.

EMP taxa were used to represent the invertebrate community as Burton et al. (2013) had already determined their sensitivity to bed sediment. The Piecewise model allowed incorporation of hierarchical/nested random structures, in this case catchment type (i.e., agricultural dairy, agricultural sheep, forestry, viticulture or reserve) nested inside coast (i.e., west or east). Triumvirate 'tests of mediation' were then used to establish which variables were significant in determining the effects of bed sediment on fish community composition. Shade and discharge were selected and incorporated in the structural equation model.

RESULTS

The mean wetted width of the first to third order waterways was 2.2 m (S.E ± 0.2), with mean water velocity of 0.24 m/s (S.E ± 0.03) and mean discharge of 0.12 m^3/s^1 (S.E ± 0.02) (Table 2.1). For water chemistry pH varied widely from 2.9 to 8.5 (Appendix E). The site which presented the lowest pH concentration (McArthur), was re-sampled the next day, after overnight rain, the pH was found to be 3.45 (Fig. 2.3) and specific conductivity of 65 µS/m. The sites location, within a remote national park, and the presence of high levels of dissolved organic carbon, observed on both occasions as clouds of bubbles on the surface, suggest natural tannins or unusual geology may be responsible for the low pH. Temperatures ranged from 9.3 – 21 °C, with a mean temperature of 13.95 °C (S.E ± 0.37). The mean dissolved oxygen 9.89 mg/l (S.E ± 0.33) while mean specific conductivity was 248.83 µS/m (S.E ± 74.50).

Strong correlations existed between all of the sediment bed cover indices, bankside assessment, quadrates and Wolman Walk (corr < 0.8450) (Appendix H).

Table 2.1 Selected characteristics of the 52 sites, including fish density, richness (number of species identified), discharge, substrate index (mean of three sediment measurements), disturbance (as indicated by Pfankuch assessment) and mean shade cover (%).

Site	Fish density (m^2)	Fish species richness	Discharge (m^3/sec^1)	Substrate index	Disturbance Pfankuch	Shade (%)
Cam	3.10	6	0.02	37.3	73	0
Fern	0.62	3	0.03	99.6	95	5
Easter	0.94	4	0.02	35.7	61	14.2
Mitre	1.92	7	0.36	6.0	76	7.4
North	0.88	5	0.36	57.3	67	16
Orton	1.28	3	0.08	7.6	82	0.4

Livings	0.69	2	0.01	63.6	85	23.6
Waipara	0.34	2	0.26	17.0	100	0
Dovetail	1.54	4	0.05	63.3	128	2.8
Jed	1.62	3	0.01	69.6	100	4.6
Benmore	2.95	3	0.01	56.6	88	1
Leader	1.40	3	0.24	17.0	107	0.2
Paurau	0.80	3	0.1	5.0	97	6.4
Fernglen	0.14	1	0.01	52.0	28	0.1
Billcliff	0.61	3	0.64	20.0	81	3.8
Okaines	3.25	7	0.14	2.7	78	13.4
Orao	0.14	6	0.21	19.3	93	0
Lyell	0.87	3	0.3	85.6	104	0
Middle	0.28	4	0.51	22.3	107	0.4
Moro	1.67	3	0.29	13.0	92	1.6
Keke	0.37	3	0.24	11.0	103	18
Acre	1.93	9	0.24	2.1	73	4.8
Braylor	1.58	4	0.11	62.6	104	13.4
Salt	1.85	4	0.01	5.0	41	16.4
Ross	5.23	8	0.02	3.5	110	0
Ward	8.85	3	0.01	66.3	122	5
Hunter	2.93	2	0.01	100.0	95	20.6
Whites	1.35	5	0.01	2.0	97	14.2
Pukana	0.85	3	0.02	14.0	77	10.6
Wilsons	1.13	6	0.02	34.8	100	4.8
Little	3.50	3	0.08	38.0	82	17.8
Kiteri	2.10	5	0.01	47.0	68	17.8
Bills	1.06	3	0.01	16.6	71	9.8
Millar	1.33	4	0.01	21.3	71	3.8
Riverside	0.41	7	0.35	9.6	103	20.6
Carey	5.02	4	0.01	23.6	77	17.2
Utopia	2.62	4	0.04	80.6	126	21.8
Cobber	1.70	5	0.02	18.6	67	18.4
Morris	2.07	5	0.01	35.6	103	5.6
Charming	0.95	5	0.02	24.4	79	11.2
Westport Wilsons	1.30	7	0.1	42.6	99	19.8
Carters	0.92	4	0.15	99.0	110	6
Serpintine	1.51	6	0.35	24.8	69	14
Waimea	0.91	7	0.53	12.5	77	6.4
Elley	1.10	2	0.01	64.0	72	18.2
9-mile	3.55	4	0.01	25.3	54	9.2
Glass-eye	6.64	3	0.01	27.3	117	18.6
Mcarthy	2.75	5	0.01	3.6	84	9.6
Birchfield	0.00	0	0.01	100.0	74	9.2
Duffys	2.83	5	0.01	78.3	114	10.8
Nikau	1.38	5	0.07	43.6	104	20.4
Barry	3.43	6	0.02	19.0	100	0

Nineteen native fishes, one introduced fish and one crustacean were captured from the 52 sites (one site had no fish). The family Galaxiidae, was represented by seven species, including all five of the migratory species that form the whitebait catch. Galaxiidae fry (whitebait) were present in 13 sites, with 149 individuals being caught in total. Two of the Galaxiidae species caught were non-migratory, Canterbury galaxias (*Galaxias vulgaris,* Stokell*)* and the Taieri flathead (*Galaxias depressiceps,* McDowall & Wallis). Each of these non-migratory species was found in only one site. The occurrence of Taieri flathead, a nationally vulnerable species was low with only nine being caught. Canterbury galaxias were more abundant with 34 individuals (Appendix I).

Figure 2.3 The sampling reach (McArthur) with the lowest pH 2.9 displayed tannin coloured waters covered in vast amounts of dissolved organic carbon.

Smelt were the most abundant with 781 individuals. Only present at three sites they formed large schools and were collected in relatively high numbers, 350 at one site (Ward) and another 292 at another site (Carey), leading to skewed total densities at those sites. However, many species were common, including several of the Eleotridae, the common bully (*Gobiomorphus cotidianus,* McDowell) occurred at 25 sites with 182 individuals. Although, the redfin bully (*Gobiomorphous huttonii,* Ogliby), was more abundant with 321 fish it was only present at 20 sites. The bluegill bully (*Gobiomorphus hubbsi*, Stokell) was present at 13 sites, with a total of 187 individuals considerably more abundant than the giant bully which was only present at 3 sites with a total of only 4 individuals caught.

Both species of eels and their elvers' occurred commonly, in a single 20 m reach there were 67 elvers caught, total abundance was 745. Longfin eels (*Anguilla dieffenbachia,* Gray) were present in 37 sites, the highest presence of any species, they were also abundant with 232 individual being caught; shortfin eels were less common with 16 presences and 109 individuals.

The families of bullies (Eleotridae) and eels (Anguillidae) were the most commonly caught, each represented by over a thousand individuals, smelt (Retropinnidae) and Galaxiidae, were also common (Fig. 2.4). However, the families Pinguipedidae, (i.e., torrentfish), Geotriidae (i.e., lamprey (*Geotria australis,* Gray), Pleuronectidae (i.e., black flounder) and Mugilidae (i.e., yellow-eyed mullet) were rare. The introduced species caught was a salmonid, brown trout, which was present at ten sites, however, nine sites contained only juveniles < 6 cm.

The density of fish captured ranged from 0 and 8.85 m^2. Average site diversity was four species, highlighting the truly depauperate nature of New Zealand waterways, even in the most pristine national park streams diversity was low, between five and nine species.

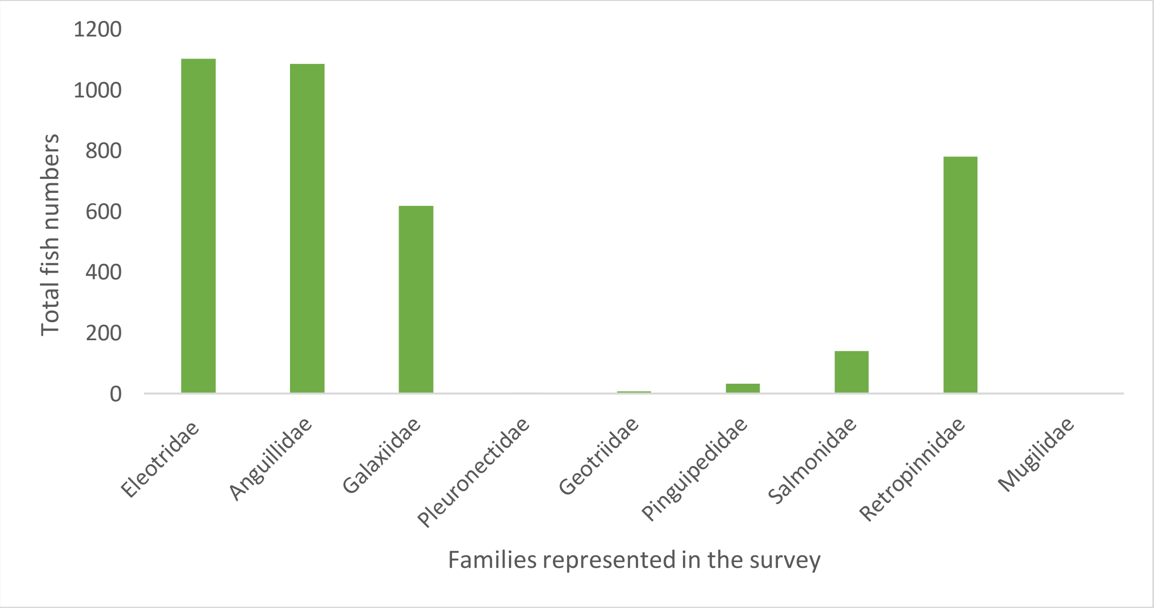


Figure 2.4 The total abundances of the nine families caught at all 52 sites. The families of Eleotridae and Anguilliade were the most common despite Galaxiidae accounting for half of New Zealand native species.

The non-linear relationship of bed sediment to density and richness

Quantile regression analysis indicated a significant non-linear, negative relationship between percentage bed sediment cover and species richness, with the upper limits set at 95% (Fig 2.5a). Although the quantile regression showed a significant negative relationship ($R^2 = 0.193$, $p = < 0.05$,), the model only explained less than 20% of the observed community change.

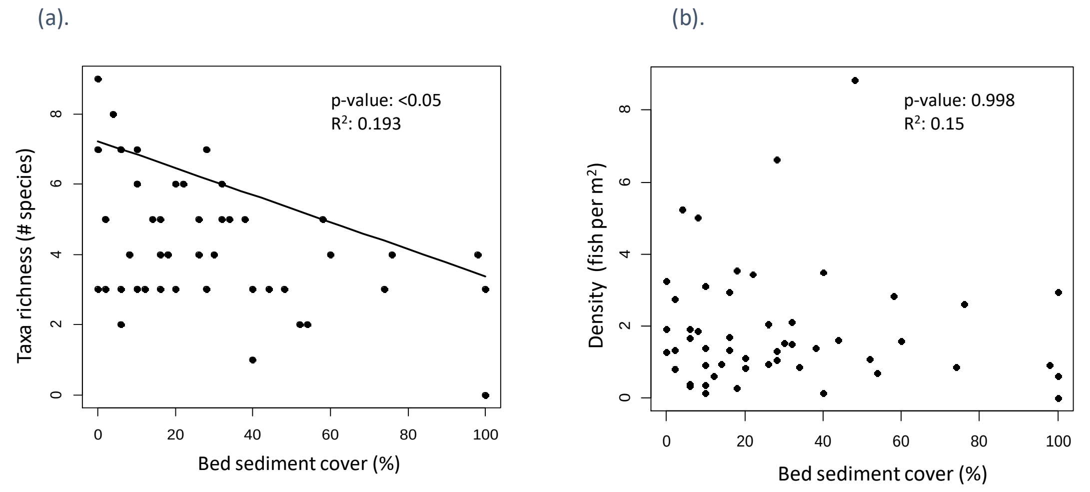


Figure 2.5 The statistically significant relationship (a) between bed sediment cover (% fines from Wolman walk) and taxa richness (the number of taxa present) across the 51 sites. However, the relationship was not significant for bed sediment cover and density. Despite the lack of a statistically significant relationship there appeared to be no occurrences of high fish abundance where there is high bed sediment cover.

Quasi-poisson regression analysis to test for relationships between percentage bed sediment cover and abundance (adjusted $R^2 = 0.015$, $p = 0.998$,) indicated that although there were negative trends the relationships were not significant (Fig 2.5b). Similarly, the relationships between bed sediment depth and both taxa richness ($R^2 = 0.01$, $p = 0.51$,) Fig 2.6a) and abundance ($R^2 = 0.015$, $p = 0.61$,) Fig 2.6b) showed only negative trends, however, they were not statistically significant.

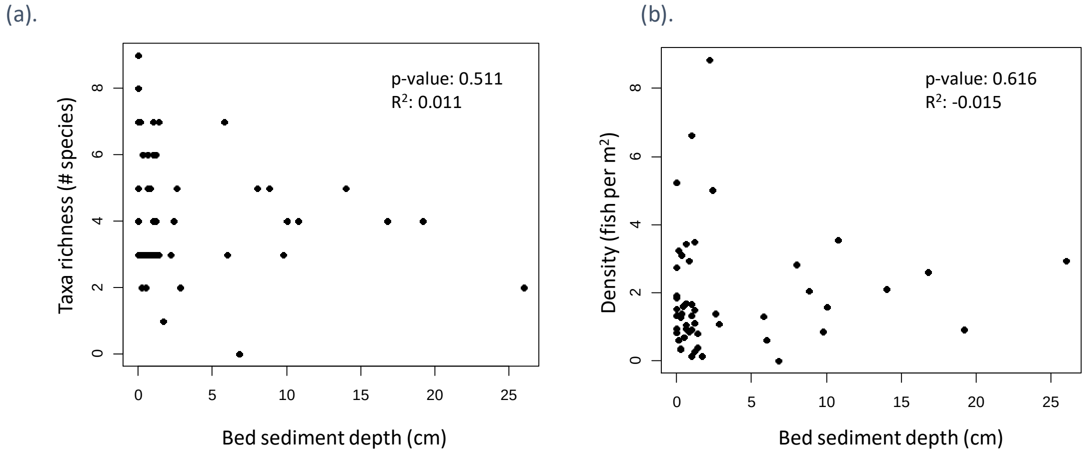


Figure 2.6 There was no statistically significant relationship between bed sediment depth (cm) and (a) taxa richness, the number of species present or (b) fish density.

Relationship between physio-chemical drivers of fish taxa richness

To determine other factors influencing fish community taxonomic composition, the data was explored with an non-metric multidimensional scaling (NMDS) ordination.

The stress value was < 0.2 (0.199), which suggested that the ordination accurately represented the dissimilarity between samples. The Shepard diagram (stress plot function, vegan Package in R) of non-metric fit illustrated that observed dissimilarities and the ordination distances were highly correlated (non-metric fit was 0.962) (Fig. 2.7)

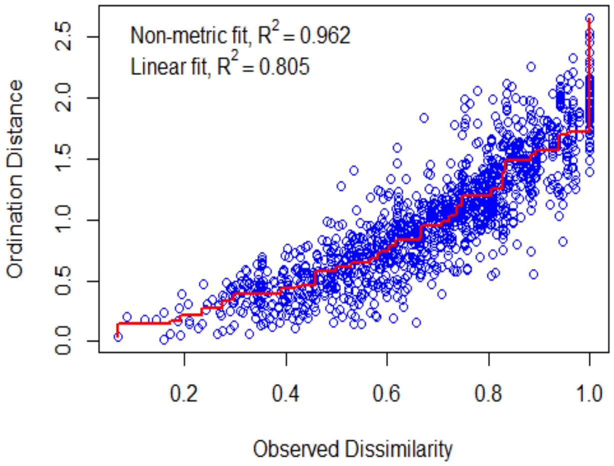


Figure 2.7 The Shepard plot, shows a the configuration of points scattered around the regression whose inter-point distances approximates the disparities, which in turn, are a nonlinear transformation of the original dissimilarities between communities.

When analysed using ordinations (NMDS) the date showed that the 52 communities were quite distinct. This wide dispersal of sites within the ordination pointed to a considerable variance between the fish communities. Variables driving the taxonomic composition of the assemblages included sediment, disturbance, discharge, salinity, shade, pH and specific conductivity. Species including the shortfin eel, upland bully and elvers were associated with higher levels of deposited sediment (Fig. 2.8) whereas species such as bluegill bully appeared to be associated with lower sediment bed covers.

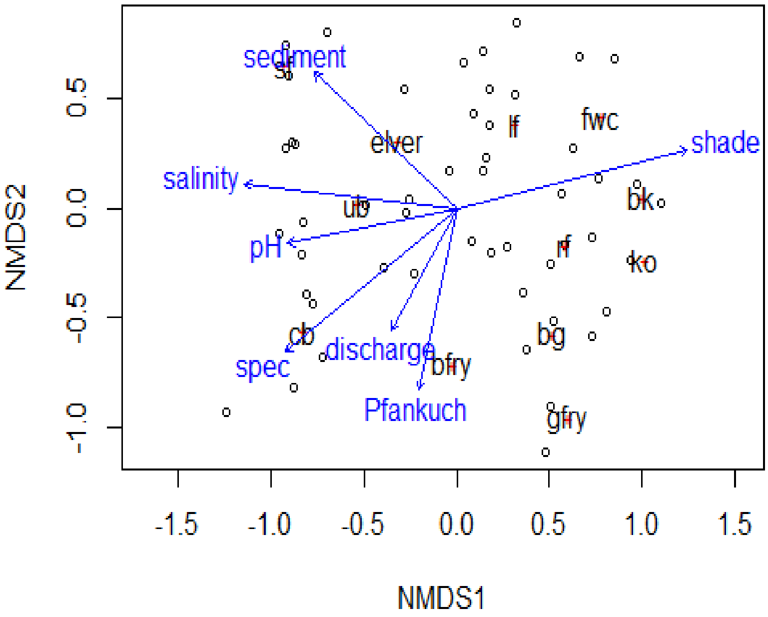


Figure 2.8 Non-metric multidimensional scaling (NMDS) ordination based on a Bray-Curtis matrix of dissimilarities calculated from fish data. The open black dotes indicated assemblages at the 51 sites (one site is not represented as fish were absent). Fish taxa as indicted in black (i.e., cb – common bully, ub – upland bully, rf – redfin bully, bg – bluegill, bfry – juvenile Eleotridae, elver – juvenile eels, lf – longfin eel, sf – shortfin eel, ko – kōaro, bk – banded kokopu, gfry – juvenile Glaxaiidae, fwc – freshwater crayfish. Variables driving the communities sampled included Pfankuch (a description of disturbance), pH, discharge, salinity, shade and sediment.

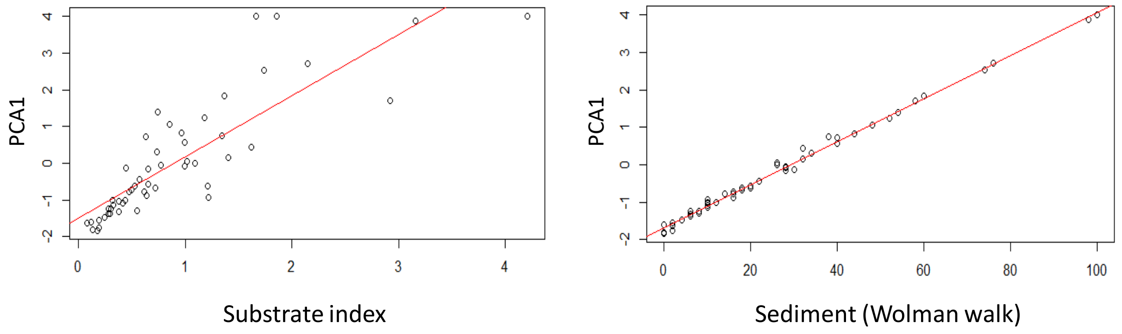
Determining the direct and in-direct drivers

Structural equation modelling (SEM) was used to determine the direct and indirect effects of selected physical (i.e., sediment, shade and discharge) and biotic variables (i.e., EPT and fish) in a causal path model. Specifically, I used SEM to investigate potential mechanistic pathways influencing fish assemblages as mediated by deposited sediment.

To construct the SEM I used principal components analysis (PCA) to decompose the substrate data into one single sediment component (PC1). The PC1 was compared with both the substrate index (the average of all three substrate variables) and percentage of sediment from the Wolman walk (Fig 2.9). It reduced sediment measurements into a single index accounting for 64% of total substrate variance.

Regression analysis was then undertaken to determine significant relations between PC1 and sediment fines ($R^2 = 0.996$, $p = < .05$,) and PC1 and the substrate index ($R^2 = 0.37$, $p = < .05$,). The chi-square (X^2) test of model fit was used to determine whether the fit between the structural equation model (SEM) and data was suitable ($P > 0.16$). The non-significant result is an indication of an adequate fit (Grace 2006). The model achieved an AIC of 43.73 suggesting a good model fit.

Analysis revealed reductions in shade were linked to increases in deposited sediment. In turn, there was a significant negative effect of sediment on invertebrates represented by EPT (%) and fish assemblages.


(a).

(b).

Figure 2.9 PC1 is compared with (a) substrate index, the average of the original three deposited sediment bed cover measurements and (b) sediment from the Wolman walk.

SEM indicated that reductions in shade, a reflection of riparian condition, contributed to increased levels of deposited sediment (Fig. 2.10). However, increasing discharge was linked to decreasing sediment (higher flow velocities leading to lower levels of deposited sediment) and increasing community richness and density, potentially as discharge is linked to habitat size.

DISCUSSION

My objective was to determine the effect of deposited sediment on fish species and density. I did this by surveying coastal, low order streams of similar size across a sediment gradient. My results showed that declines in species richness were strongly associated with increases in deposited sediment cover. When I applied regression quantiles to estimate relationships between sediment and richness, the results indicated patterns of diversity were bounded by distinct upper limits related to the controls imposed by sediment bed cover.

There is a paucity of other studies in New Zealand which have attempted to quantify the effects of sediment on fish. However, what has been published has pointed to reductions in diversity due to deposited sediment. Studies in East Coast streams of the North Island, found fish abundance and diversity reduced as sediment load increased; with up to nine fish species in low sediment sites and only two species in streams with high sediment (Richardson and Jowett 2002). This effect was not limited to the East Coast of the North Island. Similar changes to community composition were reported in the Mokau River on the North Island’s West Coast, where three species dominated sites with fine substrate (i.e., shortfin eels, common bullies and īnanga) (Hayes et al. 1989). Furthermore, Cottam and James (2003) noted in West Coast streams in the South Island, a diverse community appeared to be replaced by a community dominated by eels and common bullies, as a direct effect of sediment increases. This dominance by a few sediment-tolerant species was visible in my data. For example, at five sites with sediment bed cover exceeding 50%, the only benthic species present were eels and common bullies, often at high abundances.

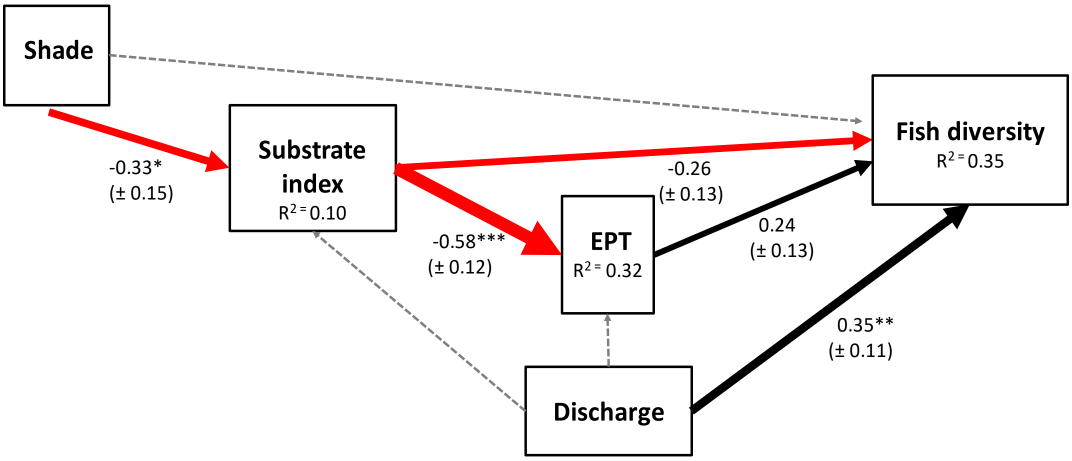


Figure 3.10 Structural equation model (SEM) showing potential causal pathways affecting fish diversity. Non-significant paths are indicated by grey dotted arrows. Black arrows denote positive relationship red arrows negative relationships. The thickness of the arrows reflects the magnitude of the standardized SEM coefficient. Bootstrapped standardized coefficients are listed beside each path, and significance levels based on a two-tailed t-statistic calculated from coefficient means. For the endogenous variables used in the model, squared multiple correlations (R^2) indicate the variance explained by all associated pathways linking that variable.

It is apparent that some of our fish species are intolerant of excessive fine sediment while others are comparatively tolerant. Sediment-tolerance in shortfin eels and elvers was apparent in my data when analysed using ordinations (NMDS). Both shortfin eels and elvers appeared to be associated with increasing levels of deposited sediment. Fish communities also changed along a salinity gradient, with salinity being linked to a sites' proximity to the ocean. The strong relationship with fish diversity and distance inland has been shown to be an important factor influencing New Zealand freshwater distributions due to diadromy (McDowall 1993). Furthermore, I found that other physio-chemical factors such as pH and specific conductivity, were important in determining the fish assemblages. The model also indicated stochastic events, such as floods, caused disturbance that influenced fish assemblages. The effect of disturbance on fish community assemblages has been seen at catchment and reach scales in both international (Resh et al. 1988) and New Zealand studies (McHugh et al. 2010).

Structural equation modelling indicated that decreased habitat availability (i.e., coarse substrate) was one of three key direct drivers of community assemblage, along with EPT (food resource) and discharge (habitat size). Habitat loss due to homogenisation has been identified as a major driver of population extinctions. In experimental channels a 60% decline in the abundance of the New Zealand upland bully, (*Gobiomorphus breviceps*, Stokell) was linked to the loss of substrate interstices (Jowett and Boustead 2001). The redfin bully, bluegill bully (*Gobiomorphus hubbsi*, Stokell) and torrentfish were shown to prefer habitats that had larger substrate and more interstitial refugia (McEwan and Joy 2014). Therefore direct effect of sediment is most likely to be linked with the reduction of habitat.

I found there was no significant link between the depth of deposited sediment and the abundance and richness of New Zealand fishes. Although, there were visible trends suggesting high taxa richness and density did not occur in assemblages at sites with deep sediment. There may be a number of possible explanations for this that I was unable to test. I would have expected that sediment cover might affect biofilm quality thereby directly impacting grazing invertebrate's food and indirectly fish. Furthermore, I would have predicted that sediment depth might be correlated with a loss of interstitial spaces, and therefore refugia from floods and predation. However, these factors might be less important in lowland streams in which food is not a limiting factor and which are often groundwater fed and not subject to severe flooding which occurs in less modified (natural) streams.

In my survey no sites with high levels of deposited sediment had high fish density, however I did not detect a significant relationship between abundance and deposited sediment. Holmes (et al 2016) indicated that fish density was not correlated with reach-scale riparian or instream habitat variables such as sediment. However, this study was undertaken in agricultural waterways where the available habitat quality gradient and fish biomass were both extremely narrow (Holmes et al. 2016). Biota in historically agricultural catchments are those that display tolerance to effects of sediment and other agriculturally derived perturbations.

The identification of specific affects, and the underlying mechanisms (e.g., loss of habitat and altered invertebrate assemblages) driving changes can only support management decisions if accompanied by thresholds. The non-linear nature of the changes observed in fish assemblages associated with increases in deposited sediment, points to the possibility of thresholds at multiple levels of organisation (i.e., community and individual taxa).

My next chapter investigates these thresholds and attempts to identify both thresholds for individual species and broader fish community thresholds.

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CHAPTER 3

DEPOSITED SEDIMENT DRIVEN THRESHOLDS FOR RIVERINE FISHES AND THE BROARDER FISH COMMUNITY

The New Zealand torrentfish, *(Cheimarrichthys fosteri*, Haast*)*, is a high-flow specialist, considered "*in decline"*.

ABSTRACT

The impacts of land-use change and intensification have had profound effects on freshwater ecosystems worldwide. In New Zealand the extensive and intensive nature of land-use changes for agriculture, urbanisation and industry have become major sources of fine inorganic sediment inputs into waterways. My earlier research found the species richness of freshwater fish assemblages was negatively impacted by increases in bed sediment. The response was non-linear indicating possible thresholds.

To determine any thresholds for individual fish species and community change, as a result of fine inorganic bed sediment, I gathered data in 52 waterways throughout the South Island of New Zealand. Application of a new analytical method, Threshold Indictor Taxa ANalysis (TITAN), allowed calculation of '*pure and reliable',* sediment thresholds for three New Zealand freshwater fish species, torrentfish (*Cheimarrichthys fosteri,* Haast), bluegill bully (*Gobiomorphus hubbsi*, Stokell) and the longfin eel (*Anguilla dieffenbachia,* Gray); as well as the Southern freshwater crayfish (*Paranephrops zealandicus,* White). Less statically robust but potentially ecologically important thresholds were determined for a number of adult native species and the juvenile stage of three families (i.e., Eleotridae, Galaxiida and Anguillidae) and the juvenile of one introduced species, brown trout (*Salmo trutta*, Linnaeus). No species showed a positive response to increasing bed sediment cover.

In an attempt to verify the TITAN results I used the thresholds generated to predict the 2011 conservation status of native species to the modelled national average for bed sediment cover. The five species considered of least conservation concern (i.e., giant bully (*Gobiomorphus gobioides,* Valenciennes), black flounder (*Rhombosolea retiaria*, Hutton), shortfin eel (*Anguilla australis*, Richardson), common bully (*Gobiomorphus cotidianus,* McDowell), and upland bully (*Gobiomorphus breviceps*, Stokell)) had sediment thresholds above the predicted national bed sediment average of 29%. Furthermore, five species considered "*in decline*", (i.e., bluegill bully, short-jaw kōkopu (*Galaxias postvectis*, Clarke), lamprey (*Geotria australis*, Gray), torrentfish, and Southern crayfish) had thresholds below the predicted national bed sediment average.

My results support a community threshold of 5-25% sediment cover at the reach scale as an appropriate target for the conservation and remediation of stream ecosystems in the New Zealand context.

INTRODUCTION

A global tragedy is unfolding in our midst. Of the 5,785 freshwater fish species assessed in 2011 by the International Union for Conservation of Nature (IUCN), more than 30% had become extinct or been identified as of conservation concern, 1,747 taxa (IUCN 2016). There are currently 15,750 described species of freshwater fishes worldwide (Carrizo et al. 2013), so there may be 4,500 freshwater fish species at risk internationally, a disappointing number.

The major global threats to freshwater fish biodiversity have been well documented and include; overexploitation, altered flow regimes (e.g., highly variable flows, low flows due to abstraction, water engineering or climate change), destruction of habitats and invasion by exotic species (Vitousek et al. 1997, Harrison and Stiassny 1999, Pimentel et al. 2005, Dudgeon. et al. 2006, Ormerod et al. 2010). These can interact with pollutants, including eutrophication and sedimentation, which influences species distribution and reduces fish diversity (Waters 1995, Seehausen et al. 1997).

The 2013 New Zealand threatened fish classification identified 74% of the 54 native freshwater fishes as threatened or at risk (Appendix A) (Goodman et al. 2014), more than double the world average (IUCN 2016). However, only a single species, the New Zealand grayling (*Prototroctes oxyrhynchus,* Günther), has been classified as extinct. This extinction occurred in the 1930's and the cause is not understood (McDowall 1990). The New Zealand Department of Conservation listed ten species as threatened in 1990; 12 years later in 2002 the number had grown to 16 species. Even now the population declines have not halted, six species had their threat status increased since 2010 because of observed declines in abundance and distribution (Goodman et al. 2014).

Despite these declines, the Freshwater Fisheries Regulation, (1983) only affectively protects introduced fish species (i.e., brown trout, rainbow trout (*Oncorhynchus mykiss*, Walbaum), chinook salmon (*Oncorhynchus tshawytscha*, Walbaum)), and the extinct native New Zealand grayling (Government 1983). In no other developed nation in the world are endangered native fish harvested commercially, culturally and recreationally. Recently (2014) the Commission for the Environment released a report stating that longfin eel and two Galaxiid species are considered under pressure from overexploitation (McDowall 1991, Environment 2014b). The endemic longfin eel, forms part of recreational, cultural and commercial catches (Jellyman 2007). Eels have traditionally been gathered for food and are considered taonga (a treasure) by Māori, therefore they may be harvested under customary rights with limited regulations (Jellyman 2007, Environment 2014b). However, commercial catch limits (20% of which are allocated to Māori), for both species of eels (longfin and shortfin), were introduced in 2000 to address sustainability concerns (Jellyman 2007). However further reductions in catch limits were necessary in 2007. The commercial fishery is currently managed using catch-limits, size limits, gear controls, area closures and voluntary measures. Whereas recreational catch is managed by the use of daily bag limits (six per person per day) and restrictions on the number of fyke nets deployed (Environment 2014a).

The other important freshwater fishery in New Zealand is the whitebait catch, made-up of the fry of five diadromous Galaxiid species (i.e., banded kōkopu (*Galaxias fasciatus*, Gray), īnanga

(*Galaxias maculatus,* Jenyns), kōaro (*Galaxias brevipinnis*, Günther), giant kōkopu (*Galaxias argenteus*, Gmelin) and short-jaw kōkopu) (McDowall 1990). Only one species, banded kōkopu, is considered not of conservation concern (Goodman et al. 2014). The whitebait fishery is managed by controlling the fishing season, gear controls and area closures (Government 1983). Despite these measures population declines continue across the country (Goodman et al. 2014), obviously either the limits are in-affectual or other factors are driving population extinctions.

In addition to the native fish, a further 23 species are either considered as colonists (3 species) or introduced and naturalised (20 taxa) (Appendix I) (Allibone et al. 2010, Goodman et al. 2014). In New Zealand the brown trout invasion is of sufficient magnitude to impact multiple trophic levels of the aquatic environment (Townsend 2003). K.R. Allen (1951) observed that invertebrate biomass in the Horokiwi Stream was actually insufficient to provide for the observed trout biomass, described as 'Allen's Paradox'. Despite some minor additions to the biomass budget (i.e., inclusion of terrestrial and drifting invertebrates) brown trout overconsumption of invertebrates (Allen 1951, Huryn 1996), poses a threat to native and endemic invertebrates. Moreover, native fish had limited experience of aquatic predators prior to the introduction of Salmonids due to New Zealand's noticeable lack of aquatic predators such as turtles, snakes and salamanders. Birds and eels prey upon all species in the family Galaxiidae, however, with relatively few predators they constituted an underexploited ecological resource. In the majority of cases where brown trout and non-migratory galaxiids compete, brown trout have displaced the Galaxiidae (Townsend 1996, 2003). Where there is co-occurrence the trout are usually small (Nilsson et al. 2006), with limited gape size, ensuring they primarily feed on invertebrates. The invasion of brown trout has been responsible for fragmentation and local extinctions of native fish populations (Townsend 1996, McIntosh et al. 2010).

Despite pressure from over-harvesting and introduced species, it is land-use changes that have had the most serious impacts on freshwater systems in New Zealand (Quinn et al. 1997). By changing waterway morphology (channelizing and damming), altering flows (increased variability or causing low flows through water abstraction) and adding contaminants (e.g., nitrogen and phosphate, heavy metals and sediment) land-use constitutes a substantial threat to biodiversity (Jellyman and Harding 2012, Environment 2013, Jellyman et al. 2013).

Water-use changes such as the demand for hydroelectric power, irrigation projects and flood mitigation schemes have encouraged widespread channelisation and dam-building. The result is that habitat is reduced, water levels and flow regimes are altered and barriers to fish migration are created (Jellyman and Harding 2012). Diadromous native fish are particularly vulnerable to migration barriers, as their distribution is controlled by their ability to penetrate inland (Hayes et al. 1989). For example, a 2012 study of 30 New Zealand dams found reduced diversity of diadromous species above dams.

However, the same study found a higher percentage of exotic species above dams when compared with numbers below dams (Jellyman and Harding 2012). Large tracts of suitable habitat can be lost because of this type of river regulation. A number of hydroelectric dams throughout the country block the migrations of diadromous species, some dams make provision for the transfer upstream of elvers, however, they do not transfer other species (NIWA 2014).

Hydroelectric dams use water resources for short periods, returning the majority to the system. However, agriculture and horticulture require large amounts of water, the majority of which is not returned, although leeching at recharge zones returns some to groundwater systems. Water for irrigation must be abstracted from aquifers (affecting springs), or from surface flows, either source will reduce flow regimes downstream. Yet, once catchments are urbanised, waterways often show more varied hydrologic regimes, as impervious surface run-off causes abrupt discharge increases during high rainfall events. The direct effects of reduced flows, caused by diversions and abstractions can be detrimental to some native species. For example, the torrentfish which is adapted to high velocity habitats can be negatively impacted when too much water is abstracted for irrigation (Hayes et al. 1989).

Reduced flows can result in higher retention times for some pollutants, especially sediment. In New Zealand, increasing sediment retention times and elevated inputs are directly linked to land-use changes, particularly agriculture, urbanisation, and extractive industries (in particular forestry, coal-mining, alluvial mining, hard rock mining, gravel extraction, china clay mining) (Davies-Colley et al. 1992, Death et al. 2003, Davie 2004, Walling 2005).

Another effect of land-use changes is the removal of riparian vegetation and narrowing of riparian buffers. This can result in bank failure and erosion (Hook 2003), directly increasing sediment loads in waterways (Jowett et al. 2009). When livestock, in particular cattle, have direct access to waterways, pugging and erosion further escalates direct sediment input (Burdon et al. 2013). Furthermore, non-point source inputs from agricultural tile drains and open drains can be additional sources of sediment (Neal et al. 2008).

The deposition, accumulation and compaction of fine sediment in the substrate, creates sediment legacies. Clapcott et al. (2011), modelled the likely streambed sediment cover for New Zealand streams prior to human colonisation, some 700 years ago. There results revealed the average streambed cover was only 8% sediment. The same models predicted the national average in 2011 to be 29% (Clapcott et al. 2011), a significant increase, which is most likely derived from anthropogenic activities, rather than natural changes to geologic patterns such as weathering and erosion.

Although it is naturally present in waterways, in both states, (i.e., suspended and deposited), excessive sediment has negative impacts on all trophic levels within an ecosystems (Danger et al. 2012, Burdon et al. 2013). In highly sediment disrupted systems, sediment-intolerant species would be expected to decline as a result of three main factors; altered food resources (i.e., quantity and quality), physical stresses (e.g., gill abrasion) and habitat homogeneity (reduces refugia).

Excessive fine sediment often fills substrate interstices decreasing the distinction between riffle, run, and pool geomorphology. As habitat heterogeneity is lost, reductions in refugia from predators can lead to increased predation (McEwan and Joy 2014b). Furthermore, the requirement to remain within a territory or particular habitat, despite currents (especially during floods), leads to an increase in energy requirements for fish lacking current refugia (Franken et al. 2006).

McDowall (2010) identified habitat homogeneity as an important factor in decreased habitat suitability for three New Zealand species, bluegill bully (McEwan and Joy 2014b), upland bully and torrentfish. This was further supported by Jowett and Boustead (2001) who showed a 60% decline in the abundance of upland bullies in experimental channels. They linked this decline to the loss of substrate interstices (Jowett and Boustead 2001). Furthermore, redfin bullies (*Gobiomorphous huttonii,* Ogliby), have been shown to prefer habitats that have larger substrate and more interstitial refugia (McEwan and Joy 2014b). This preference for interstitial refugia is possibly driven by evolutionary pressures from avian piscivorous or predatory eels.

Excessive fine sediment constitutes an anthropogenically derived stressor and it represents a novel problem for biota. At the basal level of the food web, sediment impairs both primary production (Dickman et al. 2005) and detrital breakdown (Danger et al. 2012). When detritus is smothered anoxia can occur, reducing bacterial breakdown rates. Periphyton (such as algae, cyanobacteria and other microbes) can be smothered, or have sediment incorporated into their matrix reducing their chlorophyll-a concentrations thus lowering primary productivity. This also reduces the nutritional value and can increase toxins and faecal contamination within the biofilm (Muirhead et al. 2004).

Further up the food chain, benthic invertebrate communities are negatively impacted by sediment which reduces sensitive species (i.e., Ephemeroptera, Plectoptera, Trichoptera known as EPT) (Townsend et al. 2008). A survey conducted by Burdon et al. (2013) of 30 streams in the Canterbury region focussed on sediment effects on invertebrates. The resulting structural equation models implied that the condition of the riparian margin (a proxy for land-use intensity) affected EPT invertebrate taxa via habitat change and not food availability. This relationship was driven by fine sediment inputs that buried the streambed and clogged interstitial spaces, modifying habitat and increasing homogeneity. Conversely, both Burdon et al. (2013) and Townsend et al. (2008) reported the occurrence of burrowers increased as bed sediment increased. The New Zealand aquatic snail, *Potamopyrgus antipodarum,* Gray, which is commonly found in waterways draining developed catchments with high deposited sediment, may benefit from higher sediment. When offered cobbles with diatoms or cobbles with filamentous green algae covered in sediment *P. antipodarum* chose the cobbles with filamentous green algae despite the sediment (Suren 2005).

In New Zealand streams food webs are often short, therefore predatory invertebrates, crayfish and fish, constitute apex predators. Changes to the composition of the aquatic macro-invertebrate community will alter the type of prey available to apex predators, a bottom-up affect. A bottom-up cascade occurs when a primary producer, or in this case primary consumers are removed, and there is a reduction of population size through the community (Polis and Winemiller 1996).

In an attempt to protect water quality and in some cases biotic diversity, governments worldwide are developing policies to preserve and rehabilitate rivers and streams. Most management policy guidelines relate to turbidity or suspended sediment. However, many of the negative effects of sediment on fish communities occur as a consequence of deposited sediment (Jones et al. 2012). Therefore turbidity or suspended sediment guidelines are only a partial solution to a far bigger issue. Despite a desire for conservation managers to discover contaminant thresholds very few have been identified for freshwater vertebrates. In one of the few studies, in North America, Bryce et al. (2010) calculated threshold values of 13% to protect sediment sensitive aquatic vertebrates.

In New Zealand, the government recently instituted National Policy Statement for Freshwater Management (2014). This Policy Statement established guidelines for water quality, including limits for human health (e.g., *Escherichia coli,* Migula) and nutrients (nitrates and phosphates) (Ministry for the Environment 2014). However, concerns have been raised that these will not be sufficient to protect freshwater biodiversity (Environment 2014a). For example, a measure of ecosystem health is based on nitrate toxicity levels, however, it has been suggested that the levels are currently set too low to avoid chronic ecological effects on most native fish species (Elston et al. 2015). Furthermore, despite pervasive land-use changes giving rise to sediment legacies, no guidance has yet been provided for acceptable levels of bed sediment cover (Ministry for the Environment 2014).

To facilitate the management of New Zealand freshwater fishes and protect ecosystem functions, an understanding of sediment driven thresholds for individual fish species and communities' is crucial. Thresholds are defined as "*the point at which there is an abrupt change in an ecosystem quality, property, or phenomenon, or where small changes in an environmental driver produce large responses in the ecosystem*" (Groffman et al. 2006). Understanding if these change points exist is essential as they would provide a basis from which management planning and policy can be devised (Bryce et al. 2010). However, our current knowledge of fish species and community thresholds is limited. I have addressed this lacuna by undertaking an extensive field survey of 52 lotic waterways (e.g., streams, rivers, drains and creeks).

This chapter describes the survey of first-order waterways, the goal of which was to determine the effects of deposited sediment on freshwater fish communities. The chapter provides the results of threshold analysis undertaken using Threshold Indictor Taxa ANalysis (TITAN), (Baker and King. 2010) across multiple trophic levels (i.e., individual fish species and boarder community). To further elucidate these results, I compared the thresholds derived with the 2011 national bed sediment average (Clapcott et al. 2011) and the current conservation status of freshwater fish species (Goodman et al. 2014).

METHODS

General methods

Data from the same 52 waterways sampled in Chapter 2 was used for this analysis for details of the study area refer to Chapter 2. Sediment cover in each reach was assessed by the same methods described in Chapter 2, a visual bankside assessment of percentage bed cover for the whole reach (Clapcott et al. 2011), a 50 particle Wolman walk (Wolman 1954) and measurements of sediment cover in five haphazardly placed quadrates (Clapcott et al. 2011).

Reaches were sectioned off with a 2 mm mesh stop-net at the top and bottom to create a closed system prior to fishing. This ensured most fish in the sample reach did not escape during sediment sampling.

Fishing was conducted as described in Chapter 2, with a portable (KAINGA EFM300) electric fishing machine. Each reach was fished by completing three passes of a 20 linear meter reach. Captured fish were identified in the field to species level were possible, counted and their size estimated. Very small fry (> 4 cm) were identified to family. Glass eels and elvers (> 10 cm) were recorded as elvers.

Threshold Indicator Taxa ANalysis (TITAN)

Despite numerous sediment measurement protocols being used in North America (Benoy et al. 2012) and New Zealand (Clapcott et al. 2011) there are few published studies that successfully demonstrate evidence of meaningful deposited sediment thresholds. Determining thresholds has been difficult, models often require data sets with fewer species than sampling sites, a difficult task in complex communities. Alternatively, models assume species respond in a linear fashion, however this is not necessarily the case. Responses to stressors are not invariably linear, (as described in Chapter 2) they often follow unimodal or threshold responses. Few models are able to accommodate these responses and even fewer models include rare species, despite these species often being more sensitive to stressors (Baker and King. 2010). By using TITAN I was able to included species with low abundances or few presences, however species that occurred at a single location have been excluded in this analysis as TITAN is unable to determine useful models for them.

I investigated taxon-specific and community responses, to deposited sediment using TITAN. TITAN (Appendix I), uses binary partitioning to find the bed sediment cover value that produces the greatest change in fish species abundance and occurrence across the waterways. Binary partitioning underpins statistical analyses such as classification, regression trees (Breiman 1984, De'Ath 2000) and boosted trees (De'Ath 2007).

Indicator species scores (IndVals) (Dufrêne 1997), are a highly cited method in ecological literature. TITAN uses IndVal to locate individual species change points by weighing the relative abundance of each species across candidate partitions using within-partition occurrence frequency. This provides an unbiased empirical measure of association with each side of an environmental gradient. Taxa indicator scores were scaled from 0 – 100%, with a score of 100 (perfect association), indicating the taxon was collected in every sample within a group but was absent from any other group.

Randomized permutation estimated the significance of IndVal scores. Because association was measured on either side of the partition, TITAN empirically distinguished between negative and positive species responses. To facilitate cross-taxon comparisons, taxon-specific response magnitudes were normalized (as Z scores), by comparison with values obtained from permuted samples.

In TITAN, community thresholds for increasing and declining species were estimated by summing Z- and Z+ scores separately, for each partition. The sediment gradient was partitioned to represent midpoints between every pair of successive observations with distinct environmental values. The value of sediment bed cover resulting in the largest cumulative Z scores, negative and positive, corresponds to the maximum aggregate change in a species respective frequency and abundance. These values were interpreted as a community sediment threshold if they constitute a clear maximum along the gradient, and if they occurred as the result of multiple synchronous taxon-specific change points.

TITAN estimated uncertainty surrounding thresholds using the distribution of environmental values that produce the largest sum Z- score or sum Z+ among bootstrap replicates (resampling with replacement). Bootstrapping forms the basis of random forests, boosted trees, and piecewise regression (Toms 2003). Important indicator species were identified and the significance of change assessed using indices of *purity* (consistent direction), *reliability* (consistent magnitude), and narrow quantiles (e.g., 0.05, 0.95 confidence intervals) of change-point location across the bootstrap replicates. Significance of change assessed at > 0.05 for indices of purity and reliability, however, because TITAN is not as hypothesis test, ecological important threshold values may appear at purity and reliability scores as low as > 0.08 (Appendix K). Therefore, I have included the results for individual taxa threshold but have not used them to generate the community threshold. The recommended quantile at which biological threshold should be defined is the 0.95 confidence intervals (Baker and King. 2010). TITAN was run with both 500 and 1000 iterations (bootstraps) no differences in responses were noted. Reliability and purity were reset to 0.08 allowing more species to be included in the community threshold analysis. The analysis was re-run with a small widening of the community threshold 5 – 28%.

Predicting the conservation status of New Zealand Native fish

TITAN is a new statistical approach. In an attempt to determine the accuracy of the results I compared the TITAN derived taxa thresholds with the modelled national bed sediment average in 2011. This provided insight into each taxa's potential *"sediment threat grade*", either positive or negative. Taxa with negative sediment threat grades, (those who thresholds were below the national average), would be expected to be of conservation concern. Whereas, species with positive sediment threat grades, (those who threshold were above the national average), should be of no conservation concern. Of course it is likely that harvesting and other pressures are also effecting species conservation statuses.

RESULTS

A total of 19 species were collected across the 52 sites. Detailed descriptions of the abundance and frequency of presences of the majority taxa is contained in Chapter 2. Three species (i.e., Taieri flathead (*Galaxias depressiceps*, McDowall & Wallis), Canterbury galaxias, (*Galaxias vulgaris*, Stokell) and the yellow-eyed mullet (*Aldrichetta forsteri*, Valenciennes)) did not occur at enough sites, to provide valid analysis. Longfin eels (*Anguilla dieffenbachia,* Gray) were present at 37 sites, with a total of 232 individual being caught. The bluegill bully (*Gobiomorphus hubbsi*, Stokell) was present at 13 sites, with a total of 187 individuals. Less common were the torrentfish, present in seven sites with 32 individuals and Southern freshwater crayfish present in 11 sites with 77 individuals.

TITAN was used to evaluate individual taxa thresholds to an environmental gradient; in this case the percentage of bed sediment cover. Based on species specific changes along the sediment gradient, I found three fish (i.e., torrentfish 18% (P – 1.0; R – 0.97), longfin 40% (P - 0.988; R - 0.962) and bluegill bully 24% (P – 1.0; R – 0.998) and the Southern crayfish (P – 1.0; R – 0.98) had significant purity and reliability scores (> 0.05) (Fig 3.1). These species displayed the smallest confidence intervals and strongest change point responses to sediment.

Figure 3.1 The four pure (≥0·95) and reliable (≥0·95) species are plotted in increasing order with respect to their observed environmental change points. Black symbols indicate negative indicator species, no positive indicator species (in red) were observed. The dots are sized in proportion to Z scores. Thresholds are set, and dots are placed, at the 95th confidence interval as this is seen as appropriate for biotic management (Baker and King. 2010). The red line indicates the modelled national average percentage deposited sediment (Clapcott et al. 2011).

There was a substantial difference between the thresholds of longfin 40% (P - 0.988; R - 0.962) and shortfin eels, 87 % (P - 0.91; R - 0.804). However, elvers (P - 0.624, R - 0.43) demonstrated the same threshold as adult shortfin eels. The highest threshold was also shared by the common bully.

Juvenile brown trout showed a similar threshold response to adult brown trout, at 36% and 40% respectively. However despite high purity scores (P - 0.904) brown trout juveniles showed low reliability (R - 0.572) indicating a highly negative response to sediment but with an inconsistent threshold.

Within the family Galaxiidae, banded kōkopu displayed a threshold of 41% (P - 0.926; R - 0.762). Juvenile Galaxiidae results included strong purity and reliability (P - 0.956; R - 0.942) scores with a threshold of 30%. A 59% bed cover threshold for redfin bully was unfortunately supported by low reliability score (P - 0.822; R - 0.706) that did not meet the significance criteria (Fig.3.2).

The species thresholds derived by TITAN were compared with the 2011 national average modelled by Clapcott et al. (2011). Species above and below the national average were identified. The difference between the national average in 2011 and taxa thresholds was then compared with the conservation status in 2010 (Table 3.1).

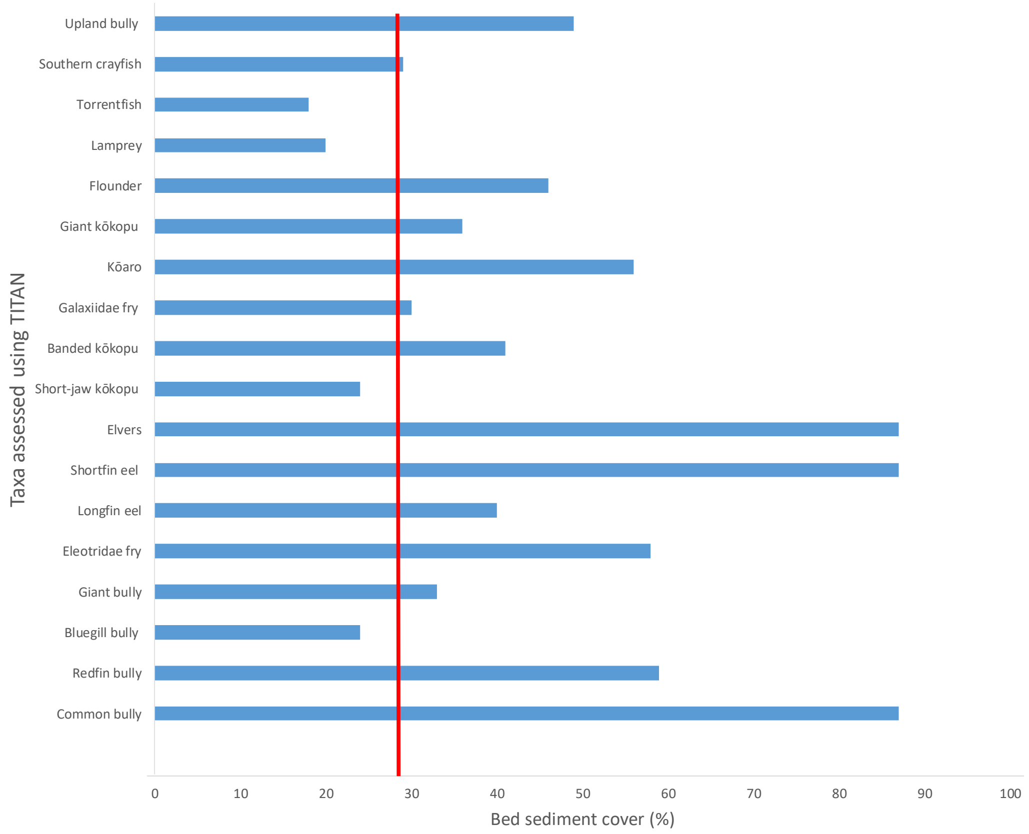

Figure 3.2 The TITAN derived threshold results for adults of 15 species and juveniles of three families at the 95% confidence interval. The red line indicates the modelled national average percentage deposited sediment (Clapcott et al. 2011).

Six species (i.e., upland bully, common bully, giant bully, shortfin eel, black flounder and banded kōkopu), had sediment thresholds above the predicted national bed sediment average of 29% (Clapcott et al. 2011), indicating they were not sensitive to the deposited sediment levels modelled for 2011. Interestingly all of these species were considered of least conservation concern in Goodman et al. (2014). Furthermore, the bluegill bully, short-jaw kōkopu, lamprey, torrentfish and the Southern crayfish are all classified as “declining”, and had thresholds below the predicted national bed sediment average of 29%. Three species declining despite displaying thresholds above the national average (i.e., kōaro, longfin eel and giant kōkopu), were recently described as under pressure from overexploitation (McDowall 1991, Environment 2014b) as they are part of commercial, recreational and cultural harvests.

Threshold analysis was then used to evaluate variation in fish community composition to percentage bed sediment cover. The cumulative response (positive or negative) of all species at a given stressor level was represented by summed Z scores. Data indicated Z- scores, however, no species displayed Z+ scores (Fig. 3.3). The greatest change in community structure, occurred between

5 - 25% bed sediment cover. This affect was the result of several species, having small confidence intervals (the associated margin of error for a given change point) and overlapping thresholds.

Table 3.2 The results from threshold indicator taxa analysis for each taxa (95% confidence interval). The taxa thresholds have been deducted from the predicted national average bed sediment cover in 2011 (Clapcott et al. 2011) defined as sediment threat grade. This provides an indication of the pressure placed on species by sediment and can be compared with the current conservation status.

Species	TITAN Threshold	Sediment threat grade	Conservation status
Upland bully	49	+20	Not Threatened
Common bully	87	+51	Not Threatened
Redfin bully	59	+30	Declining
Bluegill bully	24	-5	Declining
Giant bully	33	+4	Not Threatened
Eleotridae fry	58	+29	
Longfin eel	40	+11	Declining
Shortfin eel	87	+51	Not Threatened
Elvers	87	+51	
Short-jaw kōkopu	24	-5	Declining
Banded kōkopu	41	+12	Declining
Galaxiidae fry	30	+1	
Kōaro	56	+27	Declining
Giant kōkopu	36	+7	Declining
Flounder	46	+16	Not Threatened
Lamprey	20	-9	Declining
Torrentfish	18	-11	Declining
Southern crayfish	29	0	Declining

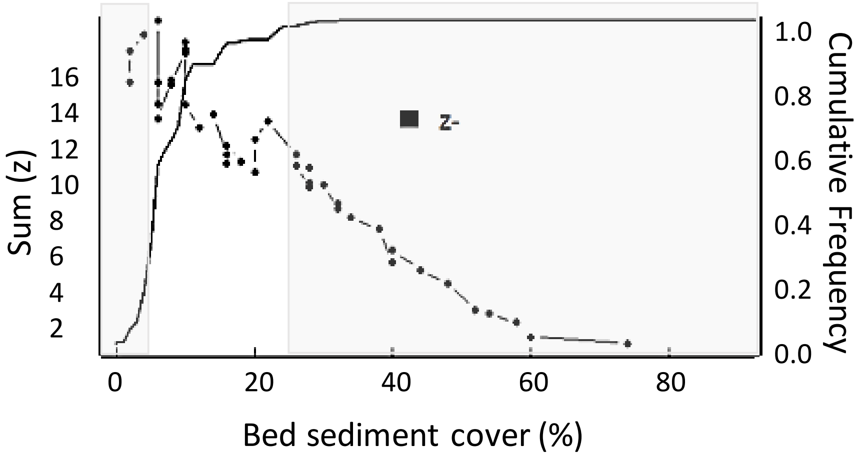

Figure 3.3 Community threshold analysis. Filtered sum Z− (black) scores, along a percentage fine sediment bed cover gradient. The bold line represents the cumulative response of all species at a given stressor level. The clear area between 5-20% (approx.) sediment cover, indicates the most community change.

DISCUSSION

In this chapter I confirmed deposited sediment was a key driver of fish species richness and showed the non-linear nature of that relationship, I further explored the survey data by undertaking thresholds analysis using TITAN. Furthermore, I was able to confirm that differing fish species were more or less sensitive to sediment and I was able to calculate sediment cover thresholds for a number of species and suggest a sediment impact range for fish communities.

The four species with the strongest threshold results (i.e., torrentfish, longfin eel, bluegill bully and Southern crayfish) response to bed sediment cover ranging from, 18 - 40%. The range of thresholds are particularly wide when the all species analysed are considered, 18 - 87%. Torrentfish displayed the lowest deposited sediment-tolerance at 18% bed sediment. This result supports McDowall's (2010) assessment that torrentfish were impacted by decreased habitat due to homogenisation and impact of increased deposited sediment. Furthermore, Hayes et al. (1989) found both torrentfish and bluegill bullies were associated with high water velocities, moderate depths and coarse substrates and in the Mokau catchment.

Habitat homogeneity was identified as an important factor in decreased habitat suitability for bluegill bullies (McDowall 2010). My threshold analysis also suggested a low sediment threshold (25%) for bluegill bullies. This is likely a reflection of their preference for habitats that have larger substrate and more interstitial refugia (McEwan and Joy 2014b) and is supported by my ordination analysis (NMDS) (Chapter 2) which placed bluegill bullies at the lower end of the sediment gradient.

Another important species, the short-jawed kōkopu, a species also classified as "*in decline*", had a sediment threshold of 24% bed cover. This species was observed to prefer streams where sediment constituted less than 10% of the substrate, whereas, it was absent at sites when sediment reached 50%. (Richardson and Jowett 2002) This observation was further supported by Cottam and James (2003) who suggested they prefer habitats with larger substrate particles and more interstitial refuge spaces when compared with all habitats available in the reach (McEwan and Joy 2014a). Both banded kōkopu and giant kōkopu had moderate sediment tolerances, 41% and 36% respectively. This may be explained by both species use of debris dams and woody debris as refugia rather than interstitial spaces (Baker and Smith 2007), however the literature on studies which have described substrate habitat preferences for these species is limited.

The 59% threshold derived for redfin bullies and upland bullies 49% are comparatively high when compared with the national average. Jowett and Richardson (2010) found redfin bullies were found among coarse substrates in larger rivers thought-out the country. However, in the Mokau catchment redfin bulliess were associated with fine substrate (Hayes et al. 1989). This observation would seem to be in contrast to my results and anecdotal observations of redfin bullies suggest they tend to occur under larger substrate and under cover (J.S. Harding *pers comm*).

The highest thresholds 87% were shared by the common bully, shortfin eels and elvers. It was unsurprising that elvers demonstrated the same threshold as adult shortfin eels, likely a result of shortfins often dominating the juvenile catch. The sediment-tolerance shown by shortfin eels and elvers was also evident in the relationship between deposited sediment and these species in my NMDS ordination (Chapter 2). High sediment-tolerance in shortfin eels has been described in various scientific literature where this species, often dominates high sediment sites (Hayes et al. 1989, Cottam and James 2003). Conversely, longfin eels displayed a comparatively low sediment threshold (40%) when compared to shortfin eels (87%). Thus, declines in habitat suitable for longfin eels driven by increased sediment cover may be actually provide opportunities for shortfin eels.

I also assessed the thresholds of juvenile brown trout which showed a similar threshold response to adult brown trout, at 36% and 40% respectively. Brown trout are known to be negatively impacted by increased sediment thought the effects on spawning redds and increases in the lethal whirling disease.

New Zealand Southern freshwater crayfish which is classified as "*in decline*", showed a low deposited sediment threshold (29%) which was equivalent to the modelled national average bed sediment cover (29%) suggested by Clapcott et al (2011), indicating the potential for further population declines in this species if sediment levels rise. Previous studies have determined the abundance of juvenile Southern crayfish was associated with coarse substrate (Usio and Townsend 2000), again pointing to habitat homogeneity as the mechanism.

Despite a wide range of tolerances no species showed a positive response to increasing bed sediment cover, suggesting there was no 'sediment community' as such, rather some species persisted despite high levels of sediment. Furthermore, comparisons between derived species thresholds, the modelled national average and conservation status indicated the TITAN thresholds may be a reasonable predictors of a species conservation status. This further implies that deposited sediment is a serious conservation concern, potentially underlying declines in species populations across the country.

Threshold analysis indicated a broader fish community thresholds of between 5 - 25% sediment covering the streambed at the reach scale, sediment bed cover exceeding this threshold will lead to less specious communities. Interestingly the community threshold is consistent with that for invertebrate community change determined as 20% by Burdon et al. (2013) on the Canterbury Plains. The threshold defined for New Zealand fish communities also encompassed the 13% threshold recommended to protect sediment sensitive aquatic vertebrates (including fish) in North America (Bryce et al. 2010).

Results of the previous invertebrate study by Burdon et al., (2013) combined with my findings, demonstrate that in the South Island a number of key species and whole communities are impacted by deposited sediment and several of these show significant threshold responses. Quantifying and understanding thresholds to deposited sediment is particularly important for the protection and rehabilitation of streams in modified catchments. These findings should help decision makers develop mitigation plans and determine rehabilitation success.

In my next chapter I investigate the impacts of suspended sediment. I attempt to determine the affect if increased turbidity to the feeding successes of four common fish species by undertaking laboratory trials.

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CHAPTER 4

VARIABLE FEEDING RATES OF FOUR NEW ZEALAND FRESHWATER FISHES IN RESPONSE TO A RANGE OF TURBIDITIES

Longfin eel (*Anguilla dieffenbachii*) on the bed of a cobbled West Coast stream, in the South Island of New Zealand. The small horn on the fish's upper-lip supports the eel's olfactory sense, which may assist the fish when foraging in turbid water.

ABSTRACT

New Zealand's geology gave rise to the evolution of diverse ecosystems with unique biodiversity. Over tens of thousands of years, New Zealand's biota including freshwater fishes, have adapted to the conditions created by active tectonics, volcanism and physical isolation. One of the South Island's most notable geologic features, the Southern Alps, is the source of both East Coast braided rivers and glacial streams on the West Coast. Braided and glacier-fed rivers, are known for naturally high turbidity levels. Turbidity can also be driven by non-geological factors, such as naturally high tannins in podocarp forested streams. New Zealand's riverine fishes evolved to use the habitats available, including waterways with naturally high turbidities.

In this chapter, I provide the results of experiments testing the feeding response of New Zealand native freshwater fishes at a range of turbidities. The effects of high turbidity (up to c. 200 NTU) were assessed by short-term (24 hour) feeding experiments. The response of juvenile shortfin eel (*Anguilla australis,* Richardson), redfin bully (*Gobiomorphous huttonii,* Ogliby), Canterbury galaxias, (*Galaxias vulgaris*, Stokell) and juvenile brown trout (*Salmo trutta*, Linnaeus) were tested at five different turbidities (i.e., 0, 5, 20, 40, 200 NTUs). Fish were provided with 30 live mayflies (*Deleatidium* spp.) to feed on over the 24 hr period.

The redfin bully showed a statistically significantly reduction in the mean feeding at 200 NTU (mean mayfly consumption 26.4) when compared with 0 NTU (mean mayfly consumption 19.3). Conversely, Canterbury galaxias, showed a statistically significant increase in feeding at 200 NTU (mean 29.3) when compared with 0 NTU (mean 25.4). However, there was no significant short term effect of turbidity on the feeding of shortfin eel, or brown trout, even at 200 NTU.

These results, and those of earlier research, suggest highly variable tolerances to turbidity exist across the New Zealand freshwater fish assemblage. Variable tolerances could result from adaptations evolved as a consequence of either naturally high turbidity or nocturnality. The results of this research has implications for policy and management of turbidity, especially in naturally high turbidity waterscapes such as braided rivers, tannin filled streams and glacial rivers.

INTRODUCTION

Globally, riverine habitats vary considerably; longitudinally (Vannote et al. 1980), temporally, and geographically (Stevenson and Sabater 2010). From the clear flowing forested streams of Canada, to the subterranean rivers beneath the Philippines, variation is the norm.

Despite the high overall diversity of waterways, some parameters are naturally common. For example, many rivers display naturally high levels of turbidity, despite the miscellany of their sources, morphology and velocity. High turbidity is found in large slow moving rivers that cut thought highly erodible soils (e.g., the River Kwai in Thailand and the Solo River, in Indonesia) and those that drain alpine areas where large amounts of sediment are eroded from the landscape and washed into the rivers (e.g., Mekong River of South east Asia) (Cushing *et al.* 1995). In glacial streams, turbidity has been shown to be significantly higher than in non-glacier fed streams. For example, in North America Stevenson and Sabater (2010) found an annual average of 125 NTU (S.E. ± 128) in glacier-fed streams, considerably higher and more variable than in non-glacial streams, with an annual average 1.9 NTU (S.E. ± 3.4). This is particularly true during the high flows experienced over summer months, when thawing increases discharge and sediment transport. Yet, high turbidity is not just the result of inorganic particles. Naturally occurring tannins, formed by the decomposition of organic matter in adjoining podocarp forests, can leech into rivers altering coloration and elevating turbidity (Fig. 4.1).

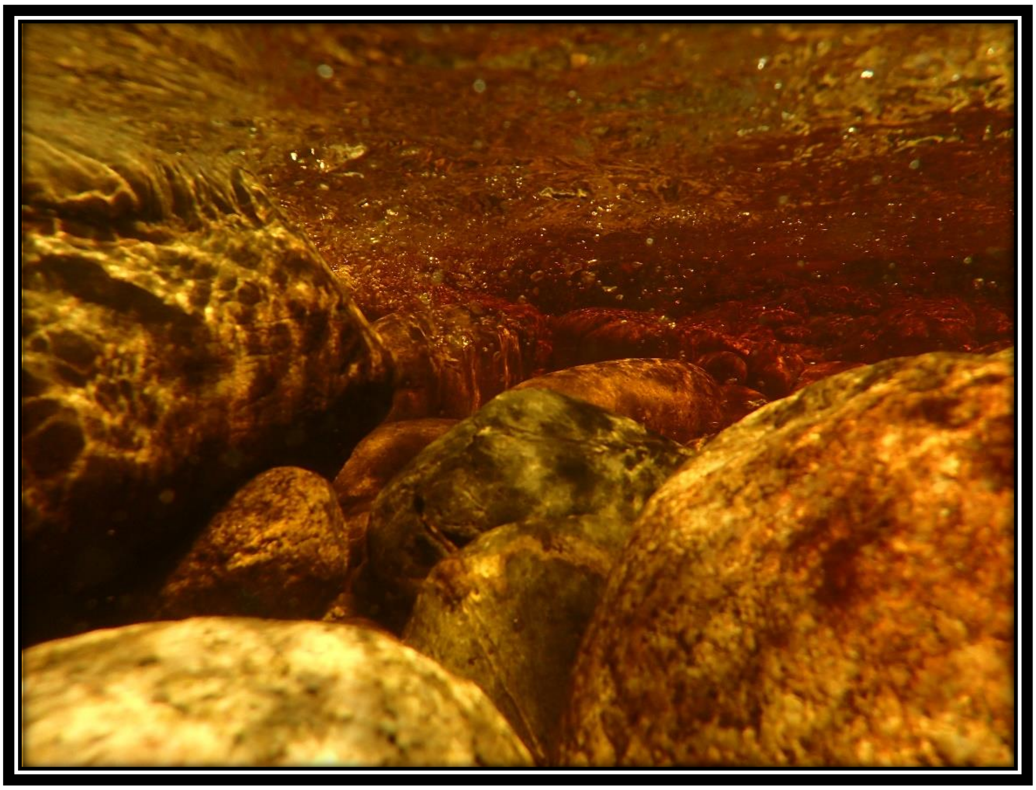

Figure 4.4 The tannin rich waters of a west coast stream within a podocarp forest, attenuates light and has naturally high turbidity levels.

Aquatic biota that inhabit naturally high turbidity waterways, have evolved to survive in the low visibility conditions experienced (Remmington 2008). Turbid water attenuates light, which negatively affects visual acuity. A loss of visual acuity, impairs the ability of predatory fish to hunt motile prey. However, many fishes have adapted to overcome reduced light. In North America, feeding trials conducted with two turbid water chub species (i.e., *Macrhybopsis tetranema*, Gilbert and *Platygobio gracilis*, Richardson), revealed both taxa were able to feed successfully at turbidities as high as 4000 NTU (Bonner and Wilde 2002). This was supported by Remmington's (2008) experiments, where turbid water species could successfully forage in turbidities as high as 3000 NTU. Even fishes that would not be considered turbid water species, such as the Tasmania golden galaxias (*Galaxias auratus*, Johnston) were able to successfully hunt in moderate turbidity. In laboratory feeding trials, planktonic and epi-benthic prey were offered to Tasmania golden galaxias, at three turbidity levels (i.e., 0, 50, 100 NTU). Neither foraging success, nor prey selection by the golden galaxias was affected by high turbidity, indicating that water clarity did not directly negatively impact feeding behaviour (Stuart-Smith et al. 2007). As a member of the family Galaxiidae, this species is closely related to more than half of New Zealand native fish fauna.

Waterscape diversity can be seen across New Zealand, a geologically young country, sited in a region of high tectonic and volcanic activity (Coates and Cox 2002). The uplifted Southern Alps are not only extremely steep, but are also prone to regularly high precipitation rates (Statistics New Zealand 2017). High rainfall increases weathering of the uplifted Alps, and their steepness advances rock abrasion. Together, the high rainfall and steep terrain support the rapid transport of eroded material into rivers and waterways. Therefore, braided rivers naturally contain high bed loads and glacial-rivers contain high wash loads (i.e., glacial flour).

The turbid environment found in these waterways requires biota to develop traits that enable existence in low light habitats. Moreover, the majority of native New Zealand fishes are nocturnal (McDowall 1990). Nocturnality demands the evolution of non-visual senses (e.g., olfactory, social, electrical and mechanosensory), often in combination with visual senses, to successfully detect and avoid predators and locate prey. Some of New Zealand's native fish use non-visual senses to navigate, detect and capture prey. As an example, torrentfish (*Cheimarrichthys fosteri,* Haast) have many superficial neuromasts which form part of a relatively well developed mechanosensory system; a development likely in response to nocturnality (Carton and Montgomery 2004). The shortfin and longfin eels are both known to use olfactory and visual cues to detect and hunt prey (Cadwallader 1973, Jellyman and Todd 1998).

In 1991, Ryan reviewed the environmental effects of sediment on New Zealand streams and stated "*Fish are not so obviously affected, although death resulting from clogging of the gills may occur in sensitive species.*" (Ryan 1991). We now know elevated turbidity can regulate behaviours in native fishes (Boubée et al. 1997, Rowe and Dean 1998), although, tolerances to turbidity vary both between species and at various phases of the life history. Rowe et al. (2000) reported that, the shortfin eels and common bullies (*Gobiomorphus cotidianus,* McDowell) were more prevalent in turbid than in clear North Island rivers. This is supported by Dean and Rowe (1998) who found that common bullies have a comparatively well-developed ability to feed using non-visual senses. Therefore, these species would be expected to be less sensitive to turbid conditions than other species. In contrast, banded kōkopu (*Galaxias fasciatus,* Gray), redfin bullies, smelt (*Retropinna retropinna*, Richardson) and īnanga (*Galaxias maculatus,* Jenyns) were considered more common in clear rivers, so are likely to be more sensitive to the effects of high turbidity conditions (Rowe *et al.* 2000).

A number of recent studies have addressed the effects of a range of turbidity levels on the feeding behaviour of New Zealand native fish, in an attempt to identify thresholds (Boubée *et al.* 1997, Rowe and Dean 1998, Richardson and Shaw 2001, Rowe and Smith 2003). Despite these studies, very few taxa have shown strong negative effects of turbidity, specifically on feeding rate. However, turbidity does impact native species in other ways (e.g., the juveniles of some species avoid turbid water when migrating).

Extremely high turbidity levels, kills fish, although notably not the New Zealand mudfish (genus *Neochanna*, Günther) which displays no negative effects of high turbidity as they aestivating when conditions become too extreme (McDowall 1990). However, for smelt once turbidity levels reach the extreme > 3,050 NTU, mortalities occurred (Hicks et al. 2009). Fortunately, such extremely high levels rarely occur in nature or, when they do, they are of short duration and therefore pose less threat. The exception may be liquefaction and volcanic ash, which once discharged may continue to increase turbidity for years with impacts on apex predators (Collier 2004, Harding and Jellyman 2015). However, the anthropogenically driven introduction of sediment into New Zealand waterways, particularly where sediment control measures and riparian management have not been effective, has led to elevated levels both a continual and episodic basis.

The aim of this chapter, was to determine if the feeding rates of four common New Zealand freshwater fishes were effected by turbidity. I tested this by a series of controlled laboratory feeding experiments. I expected increasing turbidity would reduce feeding success rates, by impairing the visual cues fish use to detect and capture prey and different taxa would show varied tolerances.

METHODS

Experiential set-up

I conducted a series of 24 hr laboratory feeding experiments on four species; adult redfin bully, Canterbury galaxias and the juveniles of brown trout and shortfin eels. These experiments consisted of placing 50 fish each into an individual tank, at one of five different turbidities, and feeding them live mayflies (*Deleatidium spp)*.

Fish were collected from the various sites (Appendix L) using a portable electric fishing machine (KAINGA EFM300). A consistent size of fish were collected for each species. This reduced unexplained variance in fish tolerance levels due to size, while ensuring normal consumption rates would be similar. Only healthy fish with no visible injuries or obvious parasites were collected.

Fish were transported in aerated containers. On arrival at the laboratory, fish were immersed for 20 minutes in a salt and bacterial treatment bath, as preliminary experiments showed fish were subject to fungal infection when stressed. The best preventative treatment was a sea salt bath at a rate of 6 g salt (NaCl) per litre water. Fish were transferred to 30 L aerated tanks with un-manipulated stream water and a flow circulation pump with aeration. Fish were acclimatised in treatment tanks for 24 hours without food prior to the commencement of feeding trials. Each tank was furnished with half a small ceramic flower pot to provide cover. All species tested use cover in the wild and were less stressed when refugia was provided.

The experiments were conducted in temperature-controlled rooms, which were maintained at 15°C with a 12 h light: 12 h dark photoperiod for the duration of the trials. This is similar to the conditions fish would encounter in the wild at the time of capture.

A total of five turbidity treatments (i.e., 0, 5, 20, 40 200), were used for each fish species. The concentrations of sediments were selected to represent control conditions (0 NTU), current Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC) (Ministry for the Environment 2000) water quality guidelines (5 NTU), and typical sediment exceedance values (20 - 200 NTU). The current ANZECC guidelines for sediment are 5.6 NTU for lowland streams and 4.1 NTU for upland streams. These values are not very realistic in the real world, as NTUs can vary by 1-2 units from a “rapid” to a “pool” a few metres away. They are also virtually impossible to achieve in a laboratory experiment. We selected 5 NTU, which is a realistic and practical concentration. Typical sediment exceedance values that occur over long time periods (> 1 week) can range from 20 – 60 NTU and many more exceedances can be > 200 NTU (J. Adams, Compliance & Consents Manager, West Coast Regional Council, *pers. comm.*). Each fish species was tested in a separate experiment; thus, four experiments were conducted.

Turbidity levels were created by adding fine porcelain clay (Halloysite). For every 5 NTU clay was added at a rate of 0.05 gm per litre. Earlier trials showed this clay was non-toxic and remained in suspension for several days when tanks were furnished with circulation pumps.

Each experiment used 50 fish, (10 fish per treatment, one fish per 30 L tank) conducted (Fig. 4.2). Ten fish were used per treatment, as previous experiments using only five or seven did not provide sufficient replication to obtain statistically significant results (Boddy & Harding, *pers. comm.*).

After the first experiments, I acclimatised the fish for an additional 24 hours prior to starting the experiment. Tanks were monitored twice per day to check fish status/condition, treatment levels (adjustments were made as required), temperature, dissolved oxygen, and ammonium concentration.

If ammonium levels reached > 1 mg L-1, then 10 mL of Stress Zyme® [i.e. a biological filter] was added to decrease ammonium levels).

Fish were fed once at the start of the experiment with 30 live mayfly nymphs (*Deleatidium*) ranging in size from 0.80 to 1.35 mm. *Deleatidium* are an invertebrate prey item commonly consumed by fish in New Zealand streams (K.L. Hogsden, unpublished data), and were collected by kick-net in the Cam River (Skewbridge), North Canterbury. Fish were offered 30 Mayflies, the lowest prey density possible because in previous experiments with higher numbers, prey density was higher than in nature, increasing the chance that fish would encounter prey despite reduced visibility.

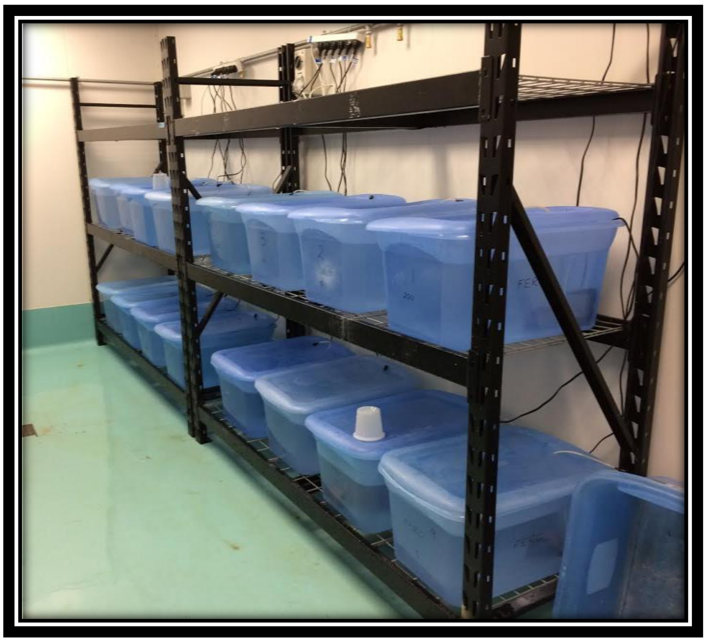

Figure 4.2 The experimental set-up with some of the 50 tanks within each of the two temperature controlled rooms. Tanks were furnished with flow pump and ceramic pot for refuge.

At the end of the experiment, fish were removed from the tanks with a small hand net and placed into a recovery tank. Water from the experimental tank was sieved through a 500-micron nytex screen kick-net and the remaining mayflies counted to determine fish consumption.

Prey control tanks were set up to ensure mayfly counts were accurate and no mayflies were damaged by flow circulation pumps.

Statistical methods

Traditional regression analysis was used to determine any significant relationships between turbidity and feeding. Statistical analysis was undertaken using R statistical software (Team 2010). All statistics were considered statistically significant where $P < 0.05$. After data sets were $\log_{10}$ transformed to ensure normality and homogeneity of variance for statistical analysis, exploratory analyses were undertaken by constructing regressions. Analyses showing data over-dispersion were re-analysed using quasi-poisson.

RESULTS

Mayfly control tanks were set up three times to ensure mayfly counts were accurate and that no mayflies were damaged by flow circulation pumps. Results showed no differences between the numbers of mayfly at the beginning of the 24 hour trial and at the end of trail. Furthermore, no mayflies were found to be damaged.

Quasi-poisson regression analysis to test for relationships between turbidity and redfin bully feeding, showed a statistically significant (adjusted $R^2 = 0.015$, $p = < 0.05$) negative result. Redfin bully had significantly lower consumption at 200 NTU compared to the control 0 NTU (Fig.4.3). However, no differences in consumption were found at 40 NTU's or less.

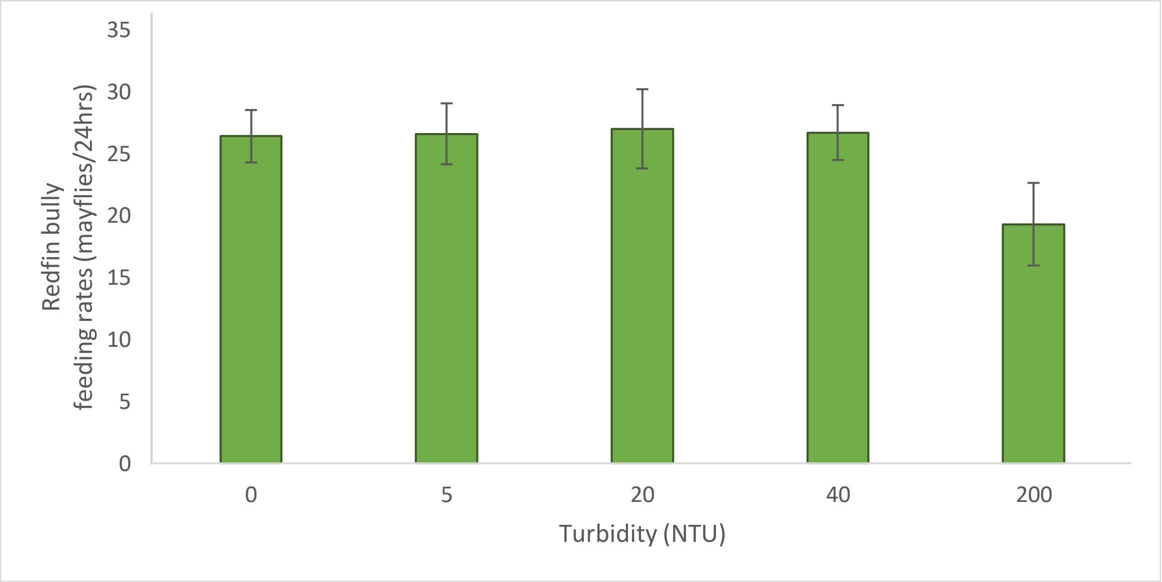

Figure 4.3 The differences in mean feeding rate of redfin bully during the 24 hour laboratory experiment. A statistically significant reduction in feeding rate occurred between 40 and 200 NTU's.

Conversely a significant positive difference was found in the mean feeding rate of Canterbury galaxias at O NTU 25.4 (S.E ± 4.3) and 200 NTU 29.3 (S.E ± 1.57). Quasi-poisson regression analysis to test for relationships between turbidity and redfin feeding showed a statistically significant (adjusted $R^2 = 0.015$, $p = 0.010$) Surprisingly, Canterbury galaxias showed an increasing consumption with increasing turbidity. At 200 NTU's seven of the ten Canterbury galaxias, consumed all of the 30 mayflies offered, two others consumed all but one mayfly at the same turbidity (Fig. 4.4).

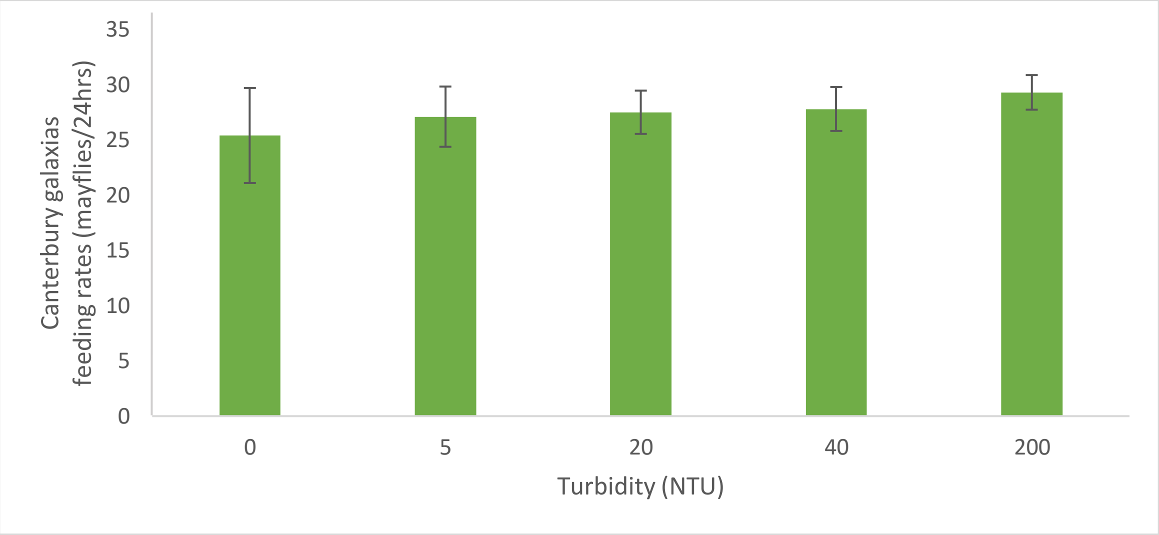

Figure 4.4 The differences in mean 24 hour feeding rate of Canterbury galaxias under the four turbidity treatments and the control. Canterbury galaxias showed a significant positive increase in feeding rate at 200 NTU.

In contrast, there were no statistically significant difference between feeding rate of juvenile shortfin eels at any turbidity (adjusted $R^2 = 0.015$, $p = 0.951$). Juvenile shortfin eels, consumed (an average of 5.04 (S.E ± 4.17) mayflies) far less overall than Canterbury galaxias (an average of 27 (S.E ± 2.87) consumed) however, several individual eels consumed no mayflies. Despite this, mean shortfin eel consumption was similar at both 0 NTU (control), 4.1 (S.E ± 4.22) and 200 NTU 5 (S.E ± 4.29) (Fig. 4.5).

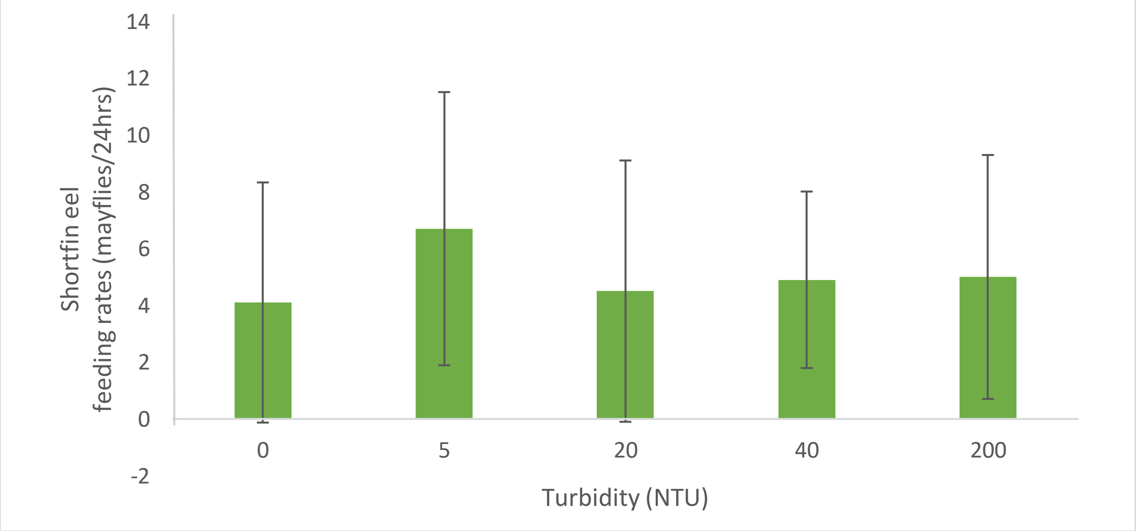

Figure 4.5 The differences in mean 24 hour feeding rate of shortfin eels under the five different turbidity levels. Shortfin eels showed high variability in feeding rate, as indicated by the magnitude of the standard errors.

Brown trout showed no statistically significant relationship between turbidity and feeding rate (adjusted $R^2 = 0.015$, $p = 0.84$), possibly due to their overall high consumption rate. Once only one or two prey remained in a tank they may have been difficult to locate even in low turbidity water. Nine of ten juvenile brown trout in 200 NTU tanks consumed more than half the prey offered, and six fish consumed all of the prey offered (Fig.4.6).

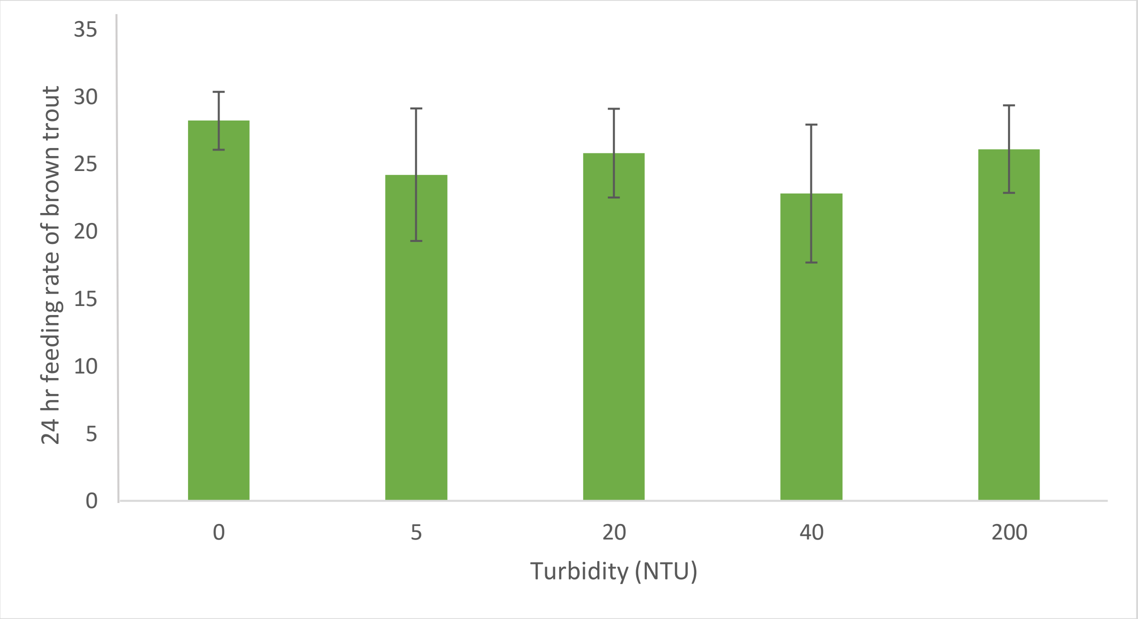

Figure 4.6 The differences in mean 24 hour feeding rate of juvenile brown trout under the four turbidity treatments and the control. Brown trout showed no significant change in feeding rate at any turbidity level.

DISCUSSION

New Zealand experiences high levels of geological activity, including rapid orogeny, volcanic eruption, faulting and earthquakes; all potentially resulting in elevated instream turbidity. The isolated nature of New Zealand's islands has provided a few freshwater fish families with opportunity to exploit all the available niches with fish even exploiting the night for foraging. The various adaptations that have evolved in response to varied turbidity levels and nocturnality have led to a range of tolerance depending on the ecological niche a taxa inhabits. For example, in choice experiments, three quarters (considered 50% avoidance) of juvenile banded kōkopu, avoided tanks with turbidity levels ≥ 25 NTU. Juvenile banded kōkopu were the most sensitive of the six migratory Galaxiidae tested. Kōaro and īnanga were less sensitive, with a 50% avoidance response at ≥70 and ≥420 NTU, respectively (Boubée *et al.* 1997).

Redfin bullies were considered by Rowe *et al.* (2000) more common in clear rivers so were likely to be more sensitive to the effects of high turbidity conditions. At 200 NTU the negative effect of turbidity on feeding rate was significant for redfin bully. However, at 40 NTU's turbidity did not have an effect of fish feeding rate, suggesting a threshold may be located somewhere between 40 and 200 NTU.

Many non-migratory species such as Canterbury galaxias may have developed a higher tolerance to turbidity than their lowland migratory counterparts. Canterbury galaxias are widely distributed on the eastern side of the South Island, including the Clarence River in the North, and extending to Trotters Gorge in the South. This extensive distribution includes many naturally turbid braided rivers and alpine streams, therefore it seems likely that Canterbury galaxias may have adapted to feed at high turbidities at least for short durations (such as 24 hours). The results of the feeding experiments indicated no sensitivity to turbidity even at 200 NTU. Increases in turbidity may even benefit Canterbury galaxias foraging rates, by attenuating light reducing the vision for invertebrate prey or by increasing their drift rate. Mayflies are a species with a high propensity to drift especially with increasing turbidity.

In tank experiments, Rowe *et al.* (2003) found that turbidity had no effect on feeding rate for rainbow trout, a Salmonid closely related to brown trout. My results, although not statistically significant, suggested that trout had low sensitivity to turbidity. Juvenile brown trout were able to consume up to 30 mayflies in a single 24 hour period, even at 200 NTU.

Shortfin eels are known to use olfactory senses to hunt prey and showed no statistically significant difference in feeding rate at any turbidity (>200 NTU). However, several eels did not consume any prey in the 24 hour period. This a common behaviour, for example adult shortfin eels at Glenshire Educational menagerie (a small zoo in Canterbury) did not feed every day, instead a single day old chicken would be sufficient for several days (3-6 days depending on the size of the eel) (Shayne Glenjarman, manager Glenshire Educational menagerie *pers. comm*.).

The varying responses of New Zealand native fish to turbidity are a likely result of the geologically driven evolution of life history patterns and diversity of riverscapes. Diadromous species often have to negotiate silted estuaries and high velocity flows containing a high bedload, glacier-fed rivers carrying glacial flour and braided rivers with their variable flows, all necessitate some level of tolerance to turbidity.

This provides some issues for policy developers, especially when setting turbidity levels with a conservation focus. In waterways where naturally high turbidities occur, fishes have evolved to manage or even benefit from the loss of visual acuity (e.g., shortfin eel, brown trout and Canterbury galaxias). However, New Zealand is not a country of solely turbid waterways, many lowland streams and forested rivers show naturally low turbidities, and of course there are fish fauna adapted to those low turbidities, for example the redfin bully.

It may be necessary to develop guidelines that take into account both continuous and episodic periods of naturally high turbidity. Precipitation driven high flows in braided rivers (e.g., Waimakariri River), and thawing in glacier-fed rivers both lead to naturally elevated turbidity. Guidelines may need to take into account the episodic nature of these events, without resorting to a blanket high turbidity threshold which could permit anthropogenically driven excesses. Conversely high tannin rivers (e.g., Black River) where turbidity is nearly constantly high will be somewhat easier to set guidelines to manage.

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CHAPTER 5

SYNTHESIS - THE WAY FORWARD

A riparian margin of mixed exotics and natives provided high quality allochthonous inputs of detritus (providing a basal resource) and woody debris for refugia. The riparian margin also offered shade which directly influenced water temperature.

INTRODUCTION

Global climate change is already placing pressure on freshwater habitats worldwide. Freshwater habitats are particularly vulnerable to climate change because both water temperature and availability are climate-dependent (IPCC 2001). Fluctuating patterns of precipitation (Vecchia et al. 2005) alter both surface flows and aquifer recharge. These changes are expected to negatively impact freshwater communities, especially apex predators, such as fish (Tisseuil et al. 2012, Comte et al. 2013, Inkley 2013). Worldwide, freshwater biota are currently disproportionally represented on the IUCN Red List (IUCN 2017). Future predictions of the decline of freshwater taxa raise serious conservation concerns which must be addressed.

The links between degradation of freshwater systems and population extinctions is strong (Dudgeon. et al. 2006). Current international literature has highlighted that excessive deposited sediment can negatively impact fish assemblages (Newcombe and Macdonald 1991, Waters 1995, Wood and Armitage 1997) by altering energy flow across trophic levels (changes to food supply) and reducing habitat heterogeneity. In the future climate change will increase sediment inputs by altering vegetation composition (Stromberg et al. 2009), increasing forest fires (Beaty 2011), and glacial recession (Micheletti and Lane 2016), in combination and more frequent extreme weather events (IPCC 2001)

In New Zealand regional studies (Cottam and James 2003, Holmes et al. 2016) offered limited insight into the effects of sediment on fish diversity and abundance. However, one recent survey of 30 streams in the Canterbury region of New Zealand, suggested that the condition of the riparian margin affected pollution-sensitive EPT invertebrate taxa via habitat change and by alterations to food quality or quantity. This relationship was driven by fine sediment inputs that smothered the streambed and clogged interstitial spaces, modifying and reducing habitat heterogeneity. The invertebrate community became dominated by sediment tolerant species once sediment bed cover exceeded 20% (Burdon et al. 2013). This is important, because such changes associated with deposited sediment may penetrate up to higher consumers (Osmundson et al. 2002). Furthermore, the same changes in taxonomic structure identified in invertebrate communities were likely to be mirrored in sediment driven changes to fish assemblages. It was on this basis I postulated that deposited sediment would cause reductions in New Zealand fish taxa richness and abundance by a combination of both reduction in energy transfer up the food web, driven by changes to the invertebrate community, and the direct effects of loss of habitat heterogeneity.

One of the other ways in which sediment can impact biotic communities is by increases in turbidity via suspended sediment. Research into the effects of turbidity (as described in Chapter 1) defined a range of different reactions from highly sensitive to highly tolerant species. In New Zealand documented changes in behaviour instigated by increased turbidity included avoidance of waterways by diadromous juveniles (Boubée et al. 1997) and reductions in foraging ability (Dean and Rowe 1998). In visual predators, foraging is effected by attenuated light. New Zealand studies have proven that turbidity affects New Zealand fishes at different life stages and at different thresholds; therefore, I expected a negative relationship between increasing turbidity and fish feeding rate, especially at very high turbidities (200 NTU). Somewhat surprisingly I found that several of the species I tested did not respond markedly to high turbidities and that in fact several were able to feed successfully in very cloudy water. Considering we have several migratory species that move during flood conditions, and we have numerous braided rivers and glacial rivers which can be naturally turbid perhaps an ability to function in turbid water is not surprising.

LOOKING BACK

In the first chapter, I reviewed the literature on the impacts of fine inorganic sediment on freshwater ecosystems. The impacts have been comprehensively described in a number of international reviews, including those by Newcombe and MacDonald in 1991 and later by Waters (1995) and most recently by Wood and Armitage (1997). Alterations in sediment provide an excellent example of how departures from natural conditions, either decreases (Wohl and Ebrary 2004, Kondolf et al. 2014) or increases in a naturally occurring subsidy, can have negative outcomes.

The most comprehensive review to date of the impacts of sediment on freshwater ecosystems in New Zealand was by Ryan (1991). However, the review only highlighted the paucity of New Zealand studies on the effects of deposited sediment on freshwater biota, especially fish. Summarily, there was a lack of information on threshold responses of individual taxa and the native fish community to deposited sediment. Unfortunately, it is this information that is necessary to inform management guidelines. Without scientifically robust guidelines, management decisions will not deliver the improvements in the water quality required by both the public and current water management policy.

Recently, the development of new sediment measurement techniques, such as bankside and bed cover assessments (Clapcott et al. 2011), have provided opportunity to further our understanding of deposited sediment.

COMMUNITY EFFECTS & STRUCTURAL FACTORS

Chapter 2 focused on determining any effect of deposited sediment on the New Zealand's fish community. An extensive survey of South Island waterways provided evidence of a statistically significant sediment induced non-linear relationship between bed sediment cover and fish diversity. Burdon et al. (2013) noted only a weak influence of deposited sediment on fish community composition. However, my research identified a statistically significant negative relationship between bed sediment and fish taxa richness. This result was not unexpected, dynamic constraints hypothesis suggested the most change within the ecosystem would be in fish assemblages, as fish represent the highest trophic level.

DRAWING THE LINE - *Winner and Losers*

In Chapter 2 the results of regression analysis identified the stressor responses evoked by sediment was not linear and potentially took the form of a threshold. In Chapter 3, I explored the concept of evolved tolerances to deposited sediment, and identified threshold responses for individual taxa and the broader community. I further explored the idea of evolved tolerance thresholds in Chapter 4 by examining fish behavioural responses to turbidity, in feeding rate experiments.

Fish responses to both deposited and suspended sediment were varied. However, most species were negatively impacted by deposited sediment (Chapter 2 and 3), whereas, turbidity responses proved to be both negative and positive (Chapter 4). The purely negative relationship between deposited sediment and diversity has ramifications for the likely composition of fish communities in sedimented systems. At high deposited sediment reaches, reductions in fish diversity were a result of the loss of sediment-sensitive species from the community assemblage. Jellyman and Harding (2015) reported extremely depauperate fish communities in Christchurch streams with increasing sediment cover resulting from earthquakes. This allowed species that are adapted to high sediment concentrations to dominate the community. For example, the bed sediment-tolerant adult shortfin eels and elvers (Chapters 2 and 3), showed no negative effects of turbidity on the feeding rate, during laboratory experiments, even at 200 NTU (Chapter 4). It is this species adaptations to sediment, such as the use other senses including olfactory to forage, that allowed shortfin eels to dominate even high bed and suspended sediment sites. Furthermore, the bed sediment threshold for redfin bully (*Gobiomorphous huttonii,* Ogliby), was indicated as 59% and laboratory experiments showed a significant negative response to turbidities between 40 and 200 NTU, both would be considered high levels of sediment. For example, the limits set by the current Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC) (Ministry for the Environment 2000) water quality guidelines suggest 5 NTU, with typical sediment exceedance values between 20 - 200 NTU.

Despite some species having high bed sediment-tolerances, in reaches were high sediment occurs, there was no species replacement only species loss (tolerant species were already present). Reductions in diversity (less specious communities) was a result of increased bed sediment cover

(Chapter 2). Species with low tolerances to both forms of sediment (suspended and deposited) may be impacted at multiple life stages by the effects of increased sediment.

Multiple effects across multiple life history phases

The combination of high levels of sedimentation and concomitant high turbidity, creates multiple stressors with the potential to work synergistically. For example, the highly valued New Zealand freshwater sport fishing industry relies heavily on brown trout. Brown trout populations may be impacted by reduction in spawning sites due to sediment infill (Wood and Armitage 1997), trout require sediment that does not exceeds 10%. However, sediment also increases the presence of the burrowing worm (*Tubifex tubifex*, Muller), which is abundant in highly sedimented sites (Brinkhurst 1996). This worm carry's a lethal disease (*Myxobolus cerebralis,* Hofer) or whirling's disease (Baerwald et al. 2011). Reductions in spawning combined with reductions in overall survival due to disease antagonistic or synergistic effects, a serious consideration for the sport fishing industry.

However, effects can be even more complex. Sediment can not only impact multiple life stages but its perturbations can be from deposited or suspended sediment. For example, brown trout, are also known to consume the majority of stream invertebrate biomass (Allen 1951). Large (>150-mm fork length) brown trout also consume most sub-adult native fish (McIntosh et al. 2010). Therefore high levels of sedimentation may change the abundance and quality of invertebrates as a food resource for brown trout and therefore, increasing their predation of native fishes such as sub-adult kōaro (*Galaxias brevipinnis*, Günther). At the same time sedimentation also reduces interstitial refugia, a habitat requirement for adult kōaro who were shown to prefer habitats with larger substrate particles (Cottam and James 2003). Furthermore, turbid river mouths inhibit the migratory colonisation of headwater streams by fry, turbidity levels ≥70 causes avoidance by migratory juvenile kōaro. It is these sorts of effects that cause population extinctions.

General community drivers

In Chapter 2 I determined the mechanisms by which sediment altered assemblages were a combination of the direct effect of modified resources (reduced EPT) and habitat (by sediment and discharge). Many previous studies have alluded to habitat homogeneity as the main mechanism by which sediment modifies taxonomic composition of fish communities and the distribution of Southern crayfish. Therefore, it was perhaps not surprising, the infilling of interstitial spaces and the burial of coarse substrate by excessive sediment has had a disproportionate effect on the primarily benthic New Zealand native fishes, that rely on this habitat to provide refugia. However, the structural equation model (Chapter 2) clearly indicated reductions in EPT were an equivalent mechanism to habitat homogenisation. Direct evidence for this process is most likely lacking, due to a paucity of research into the effect of changes to food resources on fishes.

However, theoretical studies have shown that removing the most highly connected taxa causes more cascading extinctions than random removal causes (Dunne et al. 2002). In New Zealand invertebrate grazers, shredders and scrapers are highly connected. It may be the loss of these taxa, which often belong to the families Ephemeroptera, Plecoptera and Trichoptera that is reducing fish populations. Moreover, food-web theory has shown reductions in resources caused by environmental stressors (such as sediment) allows generalists to dominate, resulting in an increase in omnivores. This would highly disadvantage New Zealand native fishes as the only omnivorous species are introduced.

Furthermore, discharge showed a positive effect on fish diversity with decreasing sediment. Discharges involvement in fish community composition could have ramifications for both water abstraction for irrigation which directly impacts downstream flows and for understanding climate change outcomes.

CLIMATE CHANGE AND SEDIMENT MECHANICS

In New Zealand, climate change is likely to drive radicle changes to water quality by the end of the decade. Current literature suggests widespread increase in river water temperatures, at a rate broadly as expected for air temperatures. A combination of reduced rainfall and an increasing requirement for water necessary for irrigation on the East Coast, where the majority of agriculture is undertaken, will decrease both peak and median flows. It remains to be seen how the interacting pressures of increasing water temperatures and reduced river flows (at least in the already drier areas on the East Coast) will further pressure water quality in waterways that are already heavily loaded by diffuse pollutants, such as sediment.

Reductions to flow velocity have been discussed previously (Chapter 1), with the two immediate impacts being increased deposition and residence time, both lead to legacy of deposited sediment. This was reflected in the structural equation model which pointed to direct links between discharge and deposited sediment. However, reductions in discharge also have direct effects on fish communities, by reducing habitat size and removing habitat for high flow specialists, such as torrentfish.

It is highly likely that the impacts of climate change will simultaneously drive up sediment inputs and reduces water velocities. The result will be an increase in the national average bed sediment cover directly posing both direct and indirect threats to fish diversity in New Zealand. The result of extremely high bed sediment was seen after the Canterbury quakes when fish communities became highly depauperate due to extreme bed sediment cover (Harding and Jellyman 2015).

Furthermore, in New Zealand diversity is almost universally found to increase when moving in a downstream direction from the headwaters; in New Zealand this is attributed the dominance by diadromous fauna (McDowall, 1993, 1996, 1998; Jowett & Richardson, 1996). Likewise, fine inorganic sediment loads tend to follow the longitudinal continuum, with inputs increasing downstream. The result is the areas with the highest biodiversity will be most affected by both sedimentation under the current climate change scenarios.

THE HERE AND NOW

In spite of the conservation concerns and the potential for worsening water quality, New Zealand native fishes lack any real effective legislative protection related to harvesting or pollutants. Therefore, they continue to face pressures mirroring those experienced by freshwater fish globally.

In truth the percentage of New Zealand native species of conservation concern far exceeds the global average. Although, New Zealand fishes face a range of anthropogenically driven impacts, from the introduction of invasive species to recreational, commercial and cultural harvesting some population declines noted can be attributed to land-use change. Furthermore, I showed some of the decline is likely directly attributed to sediment inputs, specifically deposited sediment which changes habitats. For example, all species that showed thresholds below the national average (2011) of 29% had been described as of conservation concern with notable reductions in either abundance or distribution (Fig. 5.1).

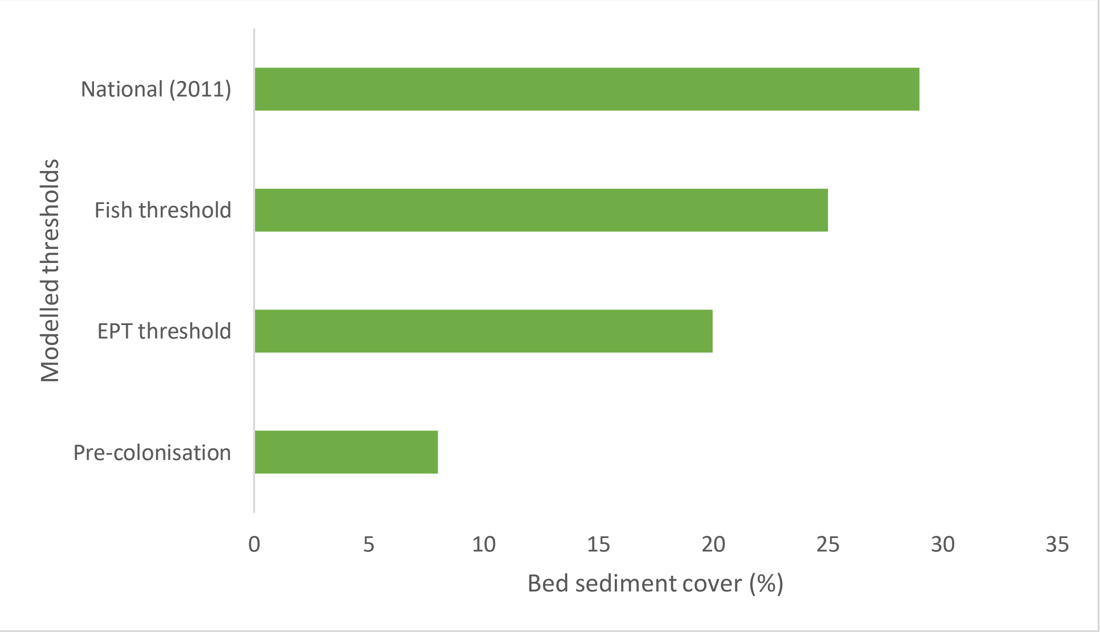

Figure 5.1 The pre-colonisation bed sediment percentage cover for hard-bottomed streams was modelled by Clapcott et al. (2011) at 8%. The same model predicted the national average sediment bed cover in 2011 was 29%. However, both the EPT taxa threshold and fish community threshold are below the national average (5 - 25%).

However, there is a bigger picture, the modelled national average is well above the pre-colonisation bed cover. Even more importantly both EPT and fish communities have thresholds below the modelled national average. This does not bode well for the 74% of fishes of conservation concern or for the endemic freshwater crayfish.

Fish populations play a role in generating ecosystem services including ecological functions and human food demands of fish. In New Zealand commercial, recreational and cultural takes are particular import as they represent tradition, economic opportunity (e.g., eel, lamprey and whitebait fishery) and to some extent food security. The majority of values placed on freshwater fishes are derived from ecosystems with complex interactions. Both economically and non-economically valuable fish populations play important roles in maintaining ecosystems services.

THE WAY FORWARD

The worldwide degradation of freshwater systems has evoked an increase in resources targeted at rehabilitating stream habitats degraded by anthropogenic activity (Lester and Boulton 2008, Roni et al. 2008, Geist and Hawkins 2016).

In New Zealand a recent change of focus, has led to more effort and resources being placed on identifying solutions rather than cataloguing the issues. Individuals, businesses, communities groups and the government are undertaking rehabilitation and restoration projects in freshwaters all over New Zealand. However, to increase restoration success, projects should be undertaken with an in-depth understanding of the biotic community’s requirements for habitat and resources. The provision of species and community thresholds allows the creation of reliable restoration goals.

Unfortunately, the effects of increased loading of fine sediment on freshwater biota (i.e., invertebrates and fish) have been identified as being driven by habitat homogeneity, a direct effect of deposited sediment. Despite this current management guidelines are based on suspended sediment targets. On this basis, existing water quality guidelines for sediment management are unlikely to be appropriate. Fortunately, there are also some well-known options in the ‘tool box’ (Monaghan et al. 2008) for control of sediment. In New Zealand most community and government sponsored rehabilitation projects focus on re-establishing riparian vegetation (Collins et al. 2013).

Here I will limit my discussion of restoration to riparian management, not because it is the primary focus of most waterways rehabilitation in New Zealand but because the structural equation model highlighted both direct and indirect links of shade on fish communities. This may be because shade is usually a good indicator of riparian quality and a healthy riparian margin has the benefits of reducing sediment inputs by stopping run-off and bank erosion (McKergow et al. 2016). My model showed reduced shade was directly implicated in increases in deposited sediment. Of course the quality of the riparian margin also controls much of the allochthonous input. Leaves which provide food resources for invertebrates and twigs and branches which contribute to refugia for some native fish species, all come from the terrestrial environment. Woody debris is considered so important for habitat heterogeneity it is being artificially added as part of restorations projects (Lester et al. 2006). A diverse riparian planting also offers the opportunity for fish to consume terrestrial insects which fall in (Inoue et al. 2013). The other positive side-effect of good quality riparian planting is the direct effect shade has on reducing water temperature (Bowler et al. 2012). Reducing water temperature not only supports fish diversity but also assists in controlling introduced macrophytes and unwanted algal blooms.

In agricultural setting land-owners have reduced inputs by excluding stock from waterways using fencing rather than riparian plantings. The potential benefits even of this simple measure are considerable (e.g., reductions in pugging and bank erosion). A significant negative correlation between the upstream area of stock exclusion fencing and deposited instream fine sediment cover exists. To achieve instream fine sediment cover below 20% in downstream reaches, 5 m wide stock exclusion fences on both banks are required (Davies-Colley 2013, Holmes et al. 2016). Furthermore, this relationship emerges when ≥300 m lengths of upstream riparian area are fenced, indicating the scale at which stock exclusion fencing results in a positive instream habitat response. However a survey of 88 riparian margins in Canterbury identified 65% ≤ 5 m wide (Renouf and Harding 2015) suggesting their bed sediment cover will exceed 20%.

Riparian margins, ideally comprised of forest plantings, can reduce sediment inputs into streams. In New Zealand, (Jowett et al. 2009) found that removal of riparian vegetation from a well-vegetated stream caused a subsequent reduction of eel density and biomass. However, reviews of the international rehabilitation literature conclude that the responses of biota to rehabilitation is often delayed or disappointing (Roni et al. 2008, Shields 2009). Although, the literature itself is somewhat scarce on long-term monitoring of rehabilitation or restoration projects. There are a multitude of reasons that fish fail to respond positively to rehabilitation efforts. The scale at which rehabilitation is undertaken is often too small, therefore it fails to address overriding catchment issues. Rehabilitation must commence at the waterways source, any issues upstream will negate any downstream efforts. Although, rehabilitation and restoration based on the premise that fish will respond positively to improved habitat diversity and water quality called the ‘field of dreams hypothesis’, (Sudduth et al. 2011) does not appear to be successful. The lack of appreciable rehabilitation success may result from the time lag between rehabilitation and fish population responses exceeding practical monitoring. Alternatively, dispersal may limit recolonization of non-diadromous species. A communities capacity to recover from sediment disturbance related loses of diversity, is dependent upon the total species pool at larger spatial scales (Fahrig 2003). In diadromous species recolonization may be limited by a lack of pheromonal ques from adults and avoidance by juveniles of high turbidity river mouths (Boubée et al. 1997, Baker and Hicks 2003). Communities may even reach a hysteresis, a situation where sediment tolerant species dominant degraded sites. However, even post restoration when habitat and food webs have been restored, the site remains dominated by a single species due to competitive exclusion.

Restoration of ecosystem function by mitigation is expensive, time consuming and has no guarantee of success. Therefore understanding sediment (i.e., turbidity and sedimentation) limits at multiple biotic levels (e.g., species, community, and ecosystem) and legislating to ensure limits are not exceeded will help protect ecosystem function and diversity and be the sounder financial option.

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Appendix A

New Zealand native and introduced freshwater fish species, threat classification status in 2009 (2008) (Goodman et al. 2014).

Common name	Binomial	Treat classification
Grayling	*Prototroctes oxyrhynchus*	Extinct
Canterbury mudfish	*Neochanna burrowsius*	Nationally Critical
Lowland longjaw galaxias (Kakanui River)	*Galaxias cobitinis*	Nationally Critical
Dusky galaxias	*Galaxias pullus*	Nationally Endangered
Eldon's galaxias	*Galaxias eldoni*	Nationally Endangered
Roundhead galaxias	*Galaxias anomalus*	Nationally Endangered
Taieri flathead galaxias	*Galaxias depressiceps*	Nationally Vulnerable
Bignose galaxias	*Galaxias macronasus*	Nationally Vulnerable
Upland longjaw galaxias (Rangitata, Rakaia Rivers)	*Galaxias prognathus*	Nationally Vulnerable
Northland mudfish	*Neochanna heleios*	Nationally Vulnerable
Gollum galaxias	*Galaxias gollumoides*	Nationally Vulnerable
Short-jaw kōkopu	*Galaxias postvectis*	Nationally Vulnerable
Lamprey	*Geotria australis*	Nationally Vulnerable
Longfin eel	*Anguilla dieffenbachii*	Declining
Torrentfish	*Cheimarrichthys fosteri*	Declining
Dwarf galaxias (West Coast)	*Galaxias divergens*	Declining
Giant kōkopu	*Galaxias argenteus*	Declining
Kōaro	*Galaxias brevipinnis*	Declining
Inanga	*Galaxias maculatus*	Declining
Bluegill bully	*Gobiomorphus hubbsi*	Declining
Redfin bully	*Gobiomorphus huttoni*	Declining
Brown mudfish	*Neochanna apoda*	Declining
Black mudfish	*Neochanna diversus*	Declining
Dwarf inanga (North Kaipara Head Dune Lakes)	*Galaxias gracilis*	Declining
Canterbury galaxias	*Galaxias vulgaris*	Declining
Tarndale bully	*Gobiomorphus alpinus*	Naturally Uncommon
Chatham Island mudfish	*Neochanna rekohua*	Naturally Uncommon
Stokell's smelt	*Stokellia anisodon*	Naturally Uncommon
Alpine galaxias	*Galaxias paucispondylus*	Naturally Uncommon
Yellow-eyed mullet	*Aldrichetta forsteri*	Not Threatened
Shortfin eel	*Anguilla australis schmidtii*	Not Threatened
Banded kōkopu	*Galaxias fasciatus*	Not Threatened
Crans bully	*Gobiomorphus basalis*	Not Threatened
Upland bully	*Gobiomorphus breviceps*	Not Threatened

Common bully	*Gobiomorphus cotidianus*	Not Threatened
Giant bully	*Gobiomorphus gobioides*	Not Threatened
Estuarine triplefin	Forsterygion nigripenne	Not Threatened
Grey mullet	*Mugil cephalus*	Not Threatened
Common smelt	*Retropinna retropinna*	Not Threatened
Black flounder	*Rhombosolea retiaria*	Not Threatened
Brown trout	*Salmo trutta*	Introduced and naturalised
Rainbow trout	*Oncorhynchus mykiss*	Introduced and naturalised
Brook char	*Salvelinus fontinalis*	Introduced and naturalised
Mackinaw	*Salvelinus namaycush*	Introduced and naturalised
Atlantic salmon	*Salmo salar*	Introduced and naturalised
Chinook salmon	*Oncorhynchus tshawytscha*	Introduced and naturalised
Sockeye salmon	*Oncorhynchus nerka*	Introduced and naturalised
Perch	*Perca fluviatilis*	Introduced and naturalised
Tench	*Tinca tinca*	Introduced and naturalised
Rudd	*Scardinius erythrophthalmus*	Introduced and naturalised
Orfe	*Leuciscus idus*	Introduced and naturalised
Goldfish	*Carassius auratus*	Introduced and naturalised
Koi carp	*Cyprinus carpio*	Introduced and naturalised
Guppy	*Poecilia reticulata*	Introduced and naturalised
Gambusia	*Gambusia affinis*	Introduced and naturalised
Sailfin molly	*Poecilia latipinna*	Introduced and naturalised
Swordtail	*Xiphophorus helleri*	Introduced and naturalised
Caudo	*Phalloceras caudimaculatus*	Introduced and naturalised
Brown bullhead catfish	*Amerirus nebulosus*	Introduced and naturalised
Glass goby	*Gobiopterus semivestitus*	Coloniser
Dart goby	*Parioglossus marginalis*	Coloniser
Australian longfin eel	*Anguilla reinhardtii*	Coloniser
Taxonomically indeterminate		
Lowland longjaw galaxias (Waitaki River)	*Galaxias aff. cobitinis 'Waitaki'*	Nationally Critical
Teviot galaxias (Teviot River)	*Galaxias 'Teviot'*	Nationally Critical
Alpine galaxias (Manuherikia River)	*Galaxias aff. paucispondylus 'Manuherikia'*	Nationally Endangered
Smeagol galaxias (Nevis River)	*Galaxias aff. gollumoides 'Nevis*	Nationally Vulnerable
Upland longjaw galaxias (Waitaki River)	*Galaxias aff. prognathus 'Waitaki'*	Nationally Vulnerable
Clutha flathead galaxias	*Galaxias sp. D*	Nationally Vulnerable
Dwarf galaxias (Nelson, Marlborough, and North Island)	*Galaxias aff. divergens 'northern*	Declining
Northern flathead (Marlborough)	*Galaxias 'Northern sp.'*	Naturally Uncommon
Dune lakes galaxias (Kai Iwi lakes)	*Galaxias sp*	Naturally Uncommon
Southern flathead	*Galaxias 'Southern sp.'*	Not Threatened

Upland bully (West Coast South Island, North Island)	*Gobiomorphus aff. breviceps*	Not Threatened

Appendix B

Table of fine inorganic sediment particle sizes and their setting velocities.

Type	Category	Particle size (mm)	Settling velocity (m's-1)
		0	
	Clay		
		0.002	
	Fine silt		
Fine sediment		0.00625>	0.0118
	Medium silt		
		0.02	0.0473
	Coarse silt		
		0.0625	0.192
	Fine sand		
		0.2	0.879
Sediment	Medium sand		
		0.63	1.76
	Coarse sand		
		2.0	2.82

Appendix C

Stocks Law and Reynolds Number, equations for determine the setting rates of small particles.

Stokes' law is given by:

$$W = \frac{2\,(Pp - Pf)gr^2}{9\mu}$$

Where:

w is settling velocity,

P is density (the subscripts p and f indicate particle and fluid respectively),

g is acceleration due to gravity,

r is the radius of the particle

μ is the dynamic viscosity of the fluid.

However, Stokes's Law (Batchelor 1967) only holds when the Reynolds number is less than one. For particles > 0.1 mm, the relationship between particle size and fall velocity is not linear due to the influence of inertia. Therefore, the Reynolds number needs to be determined and included.

To determine the Reynolds number,

$$Re = \frac{\mathrm{PvL}}{\mu}$$

Where:

P is density of the fluid

V is a characteristic velocity of the fluid with respect to the object (m/s)

L is a characteristic linear dimension (m)

μ is the dynamic viscosity of the fluid.

Other factors can influence particle deposition (i.e., particle shape, water temperature, flocculation of particles, and the turbulent nature of flow).

Appendix D

The 52 site locations with actual names and site identifiers throughout the South Island of New Zealand.

Site name	Identifier	GPS East	GPS South
Camwell Park	cam	E 171.41162	S 43.19474
Fernside Road	fern	E 172.591979	S 43.331312
Easterbrook Road	easter	E 172.584733	S 43.352063
Mitre 10 Site	mitre	E 172.596321	S 43.323683
Northbrook	north	E 172.607911	S 43.312521
Orton-Bradley Park	orton	E 172.715989	S 43.661655
Living Springs	livings	E 172.647323	S 43.645069
Waipara River	waipara	E 172.780280	S 43.074327
Dovetail Stream	dovetail	E 172.888174	S 43.097840
Jed River	jed	E 173.280832	S 42.819503
Benmore River	benmore	E 173.145840	S 42.888059
Leader River	leader	E 173.104317	S 42.603654
Puaru River	puaru	E 172.753350	S 43.647119
Fernglen Farm	fernglen	E 172.752682	S 43.647646
Billcliff's Farm	billcliff	E172.623531	S43.287772
Okaines Bay	okanies	E 173 019512	S 43 723199
Orao River	Orao	E 173.308	S 42 3053
Lyell River	lyell	E 173.4042	S 42 2302
Middle River	middle	E 173.685232	S 42.366863
Mororimu River	moro	E 173.867149	S 42.216926
Kekerengu River	keke	E 173.999909	S 41.983365
Acre Creek	acre	E 171.0636	S-42.3518
Braylor Farm	braylor	E170.5187	S 43 0736
Salt-water Forest	salt	E 170 24 33	S 43 06 21
River in Ross	ross	E 170 48 46	S 42 54 18
Ward River	ward	E 174 10 49	S 41 50 37
Hunter Road	hunter	E 173 59 21	S 41 25 51
Whites Beach	whites	E 174 05 8282	S 41 38 5416
Pukana Forest	pukana	E 173 99 7107	S 41 41 2843
Wilsons Farm	wilsons	E 173 41 32	S 41 17 12
Little	little	E 172 95 5024	S 41 08 5699
Kiteriteri Beach	kiteri	E 173 00 25	S 41 02 07
Pidgeon Valley Road	bills	E 173 01 23	S 41 22 36
Millar Creek	millar	E 170 55 7118	S 45 76 0254
Riverside Holiday Park	riverside	E170 47 36	S 45 15 11
Carey Stream	carey	E 170 34 01	S 45 42 44
Utopia Road	utopia	E 171 40 18	S 41 44 27
Fairydown Road	cobber	E 171.72 031	S 41.74 61
Morris Creek	morris	E 171.52 18	S 41.36 41
Charming Creek	charming	E 171.56 30	S 41.31 02
Westport Wilsons	westport wilsons	E 171.38 39	S 41.48 59
Carters Beach	carters	E 171.454516	S 41.890324
Serpentine Creek	serpentine	E 171.78	S 42.3443

Waimea River	waimea	E 171.0728	S 42.4031
Elley Drive	elley	E 171.55 14	S 41 75 15
9 Mile Road	nine mile	E 171.39 17	S 41.49 37
Glass-eye Road	glasseye	E 172.04 59	S 41.22 56
McArthy Road	mcarthy		
Birchfield Township	birchfield	E 171.5021	S 41.3840
Duffy Creek	duffy	E 171.5507	S 41.3317
Nikau Farm	nikau	E 171.5459	S 41.3332
Aussie Barry’s House	barry	E 171.920417	S 41.555157

Appendix E

Table of Selected physical characteristics of the 52 sites, temperature, pH, specific conductivity, dissolved oxygen (DO) in both % and mg/L, wetted width, average waterway depth, velocity and visual sediment bankside assessments of the percentage of bed sediment cover.

Site	**Tem (C^{O})**	**pH**	**Specific cond. (µS/m)**	**DO (%)**	**DO (mg/L)**	**Wetted width (m)**	**Mean velocity (m/sec)**	**Mean depth (m)**	**Sediment Bed cover (%)**
Cam	8.9	7.0	130	100	11.6	2.10	0.04	0.18	55
Fern	8.4	6.8	150	100	11.8	3.00	0.05	0.19	99
Easter	13.2	7.3	196	137	14.4	1.80	0.08	0.11	50
Mitre	10.9	7.0	129	94	10.4	2.40	0.63	0.24	10
North	13.7	6.6	106	105	10.9	3.80	0.26	0.36	80
Orton	10	7.2	144	101	11.4	3.00	0.12	0.22	15
Livings	14	8.1	436	117	12.0	0.80	0.16	0.07	55
Waipara	12.6	7.5	436	49	5.3	2.20	0.51	0.23	25
Dovetail	14.9	8.3	515	125	12.6	1.40	0.17	0.19	90
Jed	15.4	7.6	798	114	10.9	1.30	0.05	0.06	80
Benmore	16.5	8.4	332	120	11.7	1.00	0.06	0.12	75
Leader	13.3	8.5	223	110	11.4	3.90	0.41	0.15	20
Paurau	11.2	7.8	96	101	11.1	2.00	0.65	0.08	10
Fernglen	12.2	7.8	850	109	11.7	0.70	0.01	0.10	65
Billcliff	11.8	6.9	105	91	-	5.00	0.85	0.15	40
Okaines	10.5	7.6	169	102	11.4	2.60	0.33	0.17	5
Orao	13.1	7.1	171	96	10.0	6.00	0.13	0.26	20
Lyell	12.2	7.5	222	98	10.5	2.60	0.35	0.34	95
Middle	11.7	7.4	176	102	11.0	4.60	0.61	0.18	25
Moro	9.3	7.6	107	103	11.8	3.00	0.33	0.29	5
Keke	18.9	8.2	469	111	10.4	3.00	0.48	0.17	15
Acre	13.8	5.4	147	91	9.4	3.50	0.45	0.15	5
Braylor	14.9	5.8	105	84	8.5	1.80	0.18	0.33	65
Salt	12.5	3.3	34	87	9.3	1.00	0.12	0.10	5
Ross	12	6.4	52	99	10.7	1.10	0.18	0.08	5
Ward	21	7.0	3841	71	6.1	2.00	0.12	0.04	75
Hunter	15.6	7.1	152	89	8.89	2.20	0.06	0.09	100
Whites	13.6	7.3	143	98	10.2	1.00	0.18	0.07	1
Pukana	17.3	6.9	98	108	10.4	2.00	0.05	0.21	20
Wilsons	14.7	7.4	60	103	10.4	2.00	0.11	0.10	45
Little	11.4	7.1	88	99	10.8	0.80	0.54	0.19	30
Kiteri	14	6.6	73	93	9.6	0.50	0.2	0.09	25
Bills	12	6.6	77	90	9.8	2.60	0.02	0.07	15
Millar	17.3	8.1	282	108	10.4	2.00	0.03	0.11	25
Riverside	14.9	7.3	35	104	10.5	5.00	0.45	0.15	5
Carey	13.1	7.5	98	84	8.9	3.00	0.01	0.13	55
Utopia	16.5	5.9	68	85	8.3	1.30	0.23	0.13	80
Cobber	14.1	3.9	9	103	10.5	1.00	0.29	0.09	20
Morris	10.1	7.2	211	102	10.2	0.75	0.1	0.02	40

Charming	18.8	7.3	154	110	10.3	1.00	0.28	0.05	20
Westport Wilsons	14	6.1	30	104	10.7	2.50	0.26	0.16	50
Carters	15.9	5.5	132	11	1.1	3.00	0.08	0.63	99
Serpintine	16.9	6.8	52	107	10.4	4.50	0.48	0.16	35
Waimea	13.9	6.6	35	104	10.7	6.00	0.38	0.23	25
Elley	16.1	6.0	127	40	4.0	1.00	0.06	0.19	85
9-mile	13.6	6.7	47	101	10.5	1.00	0.04	0.13	30
Glass-eye	17.5	6.0	80	25	2.4	1.40	0.03	0.04	35
Mcarthy	15.1	2.9	65	101	10.1	1.00	0.05	0.11	5
Birchfield	15.9	5.3	168	77	7.7	2.00	0.01	0.30	100
Duffys	16	7.0	76	107	10.6	3.00	0.09	0.25	90
Nikau	16	6.2	107	101	10.0	0.80	0.38	0.06	40
Barrys	14.2	6.5	84.2	106	10.9	2.30	0.26	0.19	25

Appendix F

Semi-quantitative macroinvertebrate community index

Invertebrates were assessed using the semi-quantitative macroinvertebrate community index (SQMCI) (Stark 1998) a variant of the quantitative macroinvertebrate community index, however it is more cost and time effective.

$$SQMCI = \sum_{i=1}^{i=s} \frac{(n_{\mathrm{i}}\ x\ a_{\mathrm{i}})}{N}$$

where S = the total number of taxa in the sample,

n_i is the coded abundance for the *i*th scoring taxon (i.e., Rare =l, Common = 5, Abundant = 20, Very Abundant = 100, Very Very Abundant = 500),

a_i is the biotic index score (e.g., 1 for pollution resilient taxa, 9 for pollution sensitive taxa) for the *i*th taxon,

N is the total of the coded abundances for the entire sample.

Appendix G

Carle-Strub depletion estimator

From quantitative electrofishing results, I estimated the abundance of each taxa by maximum weighted likelihood with the Carle-Strub depletion estimator.

$$T = \sum_{i-1}^{k} Ci$$

$$X = \sum_{i=1}^{k} (k-1)Ci$$

$$\left[\frac{n+1}{n-T+1}\right] \prod_{i-1}^{k} \left[\frac{kn - X - T + 1 + (k-i)}{kn - X - 2 + (k-1)}\right] i \leq 1.0$$

where, i = pass number,

k = number of removals (passes),

C_i = number of fish caught in i^{th} sample,

X = an intermediate statistic used above,

T = total number of fish caught in all passes.

n iterative process by substituting values for n until

The numbers of fish per cubic metre of channel (fish m^3), were calculated as the product of fish density (fish m^2) and wetted width (m). Total fish abundances were calculated as fish per m^3 for each taxa then summed to give the total fish density for each site.

$$F = D * w$$

where, F = fish m^3,

D = fish per m^2

w = wetted width

Appendix H

Correlations between biotic and sediment variables

	Reach	**Wolman**	**Quadrates**	**Index**	**Sediment volume**	**Sediment depth**
Abundance	0.05	-0.07	-0.10	-0.04	-0.01	-0.01
Fish per m³	0.03	-0.01	-0.01	0.01	-0.03	0.01
Richness	-0.44	-0.45	-0.40	-0.02	-0.20	-0.17
Natives	-0.43	-0.43	-0.38	-0.43	-0.17	-0.13
Reach	1	0.84	0.87	0.95	0.56	0.53
Wolman	0.84	1	0.85	0.94	0.75	0.71
Quadrates	0.87	0.85	1	0.96	0.66	0.63
Index	0.95	0.94	0.96	1	0.69	0.65
Sediment volume	0.56	0.75	0.66	0.69	1	0.96
Sediment depth	0.53	0.71	0.63	0.65	0.96	1

Appendix I

The number of sites each taxa was present, the percentage of occurrences in the 51 sites and the total abundance of each taxa.

Species	Number of sites present	% occurrence	Total caught
Eleotridae (5 species)			1102
Gobiomorphus breviceps Upland bully	11	43	115
Gobiomorphus cotidianus Common bully	25	49	182
Gobiomorphus huttoni Redfin bully	20	39	321
Gobiomorphus hubbsi Bluegill bully	13	25	187
Gobiomorphus gobioides Giant bully	3	6	4
Eleotridae fry	18	36	293
Anguillidae (2 species)			1086
Anguilla dieffenbachii Longfin eels	17	33	232
Anguilla australis Shortfin eels	16	31	109
Elvers	47	92	745
Galaxiidae (7 species)			619
Galaxias vulgaris Canterbury galaxias	1	2	34
Galaxias postvectis Short-jawed kokopu	3	6	12
Galaxias fasciatus Banded kokopu	14	27	70
Galaxias depressiceps Taieri flathead	1	2	9
Galaxias brevipinnis Koaro	14	27	142
Galaxias argenteus Giant kokopu	3	6	3
Galaxias maculatus Inanga	15	29	200
Galaxia fry	13	25	149
Pleuronectidae (1 species)			4
Rhombosolea retiaria Black flounder	3	6	4
Geotriidae (1 species)			8
Geotria australis Lamprey	3	6	8
Pinguipedidae (1 species)			32
Cheimarrichthys fosteri Torrent fish	7	14	32
Salmonidae (1 species)			141

Salmo trutta Brown trout	6	12	19
Juvenile brown trout	5	10	122
Parastacidae			77
Paranephorus. zealandicus Koura (crayfish)	11	21	77
Retropinnidae (1 species)			781
Retropinna retropinna Common smelt	3	6	781
Mugilidae (1 species)			1
Aldrichetta forsteri Yellow-eyed mullet	1	2	1
Total			3851

Appendix J

Threshold Indicator Taxa ANalysis (TITAN) (Baker and King. 2010)

TITAN identifies thresholds, biological transition points or zones of rapid change in a community which occur in response to small, continuous increases in a stressor. To achieve this there are five main manipulations;

1. Preliminaries
    a) Order the 51 samples units along the predictor variable *x* (often referred to as the environmental gradient in this case sediment).
    b) Identify midpoints between unique values of x as potential change points x_i.
2. IndVals for each candidate change point
    2.1 For each species
        a) IndVal scores are calculated from samples grouped above and below each possible change point x_i. IndVal scores estimate the association of each species with each group. Thus, two IndVal scores are computed for a single species in a two-group classification. IndVal scores are the product of cross-group relative abundance and within-group occurrence frequency.

$IndVal = a \times b$

Where

a is the proportion of abundance among all sample units belonging to *group i*

b is the proportion of sample units in *group i* with a positive abundance value

IndVal use within group presences data to weight a species' relative abundance by how consistently it is observed within a group. A species with high abundance within a group will have a high IndVal score however only if the species is present in several samples within the group. Because IndVal scores integrate both occurrence frequency and abundance they provide a measure of association that is unbiased by group size (Dufrêne and Legendre 1997).

IndVal scores are scaled from 0% - 100%. A value of 100% would indicate a species was collected in all samples within a group and was absent of all other groups. In a two group classification, a value of 50% indicates a species occurred in equal abundances in all samples of both groups or in 50% of the samples in only one group.

A IndVal p value is derived by comparing the magnitude of each observed IndVal score with those generated when group membership is assigned via ≥250 randomized permutations (Dufrêne and Legendre 1997).

b. Compare IndVals above and below each potential change point $x_{i,}$ retain the higher score.

c. Identify maximum IndVal across x_i, the observed change point x_{cp}, is the corresponding value of x.

d. Assign species to positive or negative response groups.

2.2 Repeat this process with ≥250 iterations reshuffling x_i to estimate the likelihood of obtaining IndVals ≥ the observed maximum IndVal (p) and the mean and standard deviation of random IndVals.

3. Community thresholds from multiple species change points.

a. Standardise observed IndVals as *z scores* using the mean and standard deviation of IndVals.

$$z\ score = \frac{\text{IndVal} - \text{mean}}{\text{Standard deviation}}$$

b. Sum species *z scores* by response group assignment (*-ve* or *+ve*) for each potential change point x_i.

a. Identify x values corresponding to sum *(-z* and *+z*) maxima as community thresholds.

4) Bootstrap estimate of uncertainty (Sampling with replacement)

a. Repeat the process with a minimum of 250 bootstrap iterations.

5) Diagnose and Interpret

a. Use bootstrap data to estimate confidence limits for summed *-z* and *+z*, and species change points.
b. Calculate *purity* and *reliability* for each species. *Purity* (P) is the proportion of bootstrapped replicates whose group assignment matched observed assignment. *Reliability* (R) is the proportion of replicates whose maximum IndVal ≥ p. The user specified α level was defined as 0.05.
c. Define indicator species using indicators of purity, reliability and width of confidence levels, use these indicator species to interpret community thresholds.

Appendix K

Results from threshold indicator taxa analysis including IndVal, z scores, confidence intervals from 5-95% (the species thresholds being set at the 95 %), z median, purity and reliability scores for the adults of 16 fish taxa the juveniles of three families and the Southern crayfish. *** denotes species with both purity and reliability which are significant at < 0.95 ** denotes species with purity and reliability which are ecologically important at < 0.80

Species	IndVal	Z score	5%	10%	50%	90%	95% threshold	Purity	Reliability	z.median
Brown trout	17.47	1.35	1	2	19	33	**40**	0.696	0.434	2.097773
Brown trout fry	19.84	1.82	6	7	13	28	**36**	0.904	0.572	2.249378
Upland bully	25.58	1.08	8	9	19	42	**49**	0.598	0.53	2.128527
Common bully	32.43	0.15	1	1	20	68.6	**87**	0.684	0.554	2.186274
Redfin bully	42.37	1.43	1	1	10	42	**59**	0.822	0.706	2.605543
Bluegill bully ***	61.27	5.19	1	2	7	24	**24.05**	**1**	**0.998**	6.627033
Giant bully	8.11	0.4	6	6	19	33	**33**	0.492	0.182	1.409225
Eleotridae fry	31.23	0.74	7	8	31	56	**58**	0.544	0.5	2.010717
Longfin eel ***	83.45	4.22	2	3	5	30	**40.1**	**0.988**	**0.962**	4.120903
Shortfin eel **	50.74	3.54	13	26.9	54	87	**87**	**0.91**	**0.804**	3.829913
Elvers	54.11	0.42	4.95	10	31.5	75	**87**	0.624	0.43	1.772587
Short-jaw kōkopu	39.62	6	2	2	4	21	**24**	0.876	0.586	4.777223
Banded kōkopu	39.43	1.74	3	4	21	39	**41.05**	0.926	0.762	3.002907
Galaxiidae fry **	56.74	5.15	5	6	7	11.1	**30**	**0.956**	**0.942**	4.759813
Kōaro	30.69	0.68	2	2	17.5	54	**56**	0.644	0.494	1.931367
Giant kōkopu	10.9	0.38	5	5	21.5	35	**36**	0.46	0.224	1.863052
Flounder	8.11	0.52	10	10	20	46	**46**	0.492	0.198	1.4973
Lamprey	12.5	1.78	4	6	18	20	**20**	0.824	0.296	1.931911
Torrentfish ***	51.35	6.29	4	5	6	11.1	**18**	**1**	**0.97**	7.196294
Southern crayfish ***	42.97	3.92	3	3	10	23	**29**	**1**	**0.98**	4.792341

Appendix L

The four species assessed during the laboratory experiments, the co-ordinates of sites were fish were collected and the range of sizes of the fifty fish of each species.

Common name	Scientific (binomial) name	Site GPS co-ordinates	Size range
Redfin bully	*Gobiomorphous huttonii*	-42.653788, 173.266464	4 – 5 cm
Shortfin eel	*Anguilla australis*	-43.776177, 172.359597	10 – 20 cm
Brown trout	*Salmo trutta*	-43.131388, 171.766690	10 – 15 cm
Canterbury galaxias	*Galaxias vulgaris*	-42.653788, 173.266464	7 – 9 cm

PERMITS AND CONSULTATION

The research was conducted under Animal Ethics Permit 2015/OR5 issued by the University of Canterbury. Drugs were used under IDAO 05-15/01 – the pertinent “drugs” being Aqui-S and Stress zyme. The Department of Conservation (DoC) provided an electric fishing permit and permits to survey 187 waterways around the South Island wide. Consultation with Māori was undertaken by DoC as per their permit process. Land owners were contacted prior to entry of private property.

PRECAUTIONS

After each reach was sampled all equipment was cleaned and dried in compliance with didymo spread prevention methods. Sites were selected to ensure staff safety.