The application of ecological theory to the remediation of macroinvertebrate communities impacted by acid mine drainage

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Abstract

Numerous streams on the West Coast drain catchments impacted by active or abandoned coal mining areas. Acid mine drainage (AMD) from coal mining can have significant negative effects on stream communities. Changing environmental ethics and regulations mean that mining companies are now encouraged to treat acid mine drainage to enable streams communities to recover. However, remediation efforts have not always been ecologically successful, and mining companies are seeking methods to enhance macroinvertebrate community recovery. Initially, I conducted an extensive survey of 45 streams draining the Stockton Plateau, which is the site of the largest opencast coal mine in New Zealand. I assessed physical and chemical conditions at each site as well as sampling benthic communities. This spatial survey showed streams impacted by acid mine drainage were comprised of chironomids and AMD-tolerant caddisflies such as *Psilochorema* and stoneflies such as *Spaniocercoides*. Un-impacted streams typically had a pH of ~5 and were dominated by mayflies (*Deleatidium* or *Zephlebia*). Analysis revealed that stream location within the landscape also had a significant influence on macroinvertebrate community composition. Another aspect of stream recovery is the ability of species to recolonise a stream. Therefore, I investigated the flight direction of adult aquatic insects in order to determine longitudinal and lateral flight preferences. No significant differences in flight direction were observed. I also investigated the influence of riparian habitat on lateral dispersal and found that a number of patterns were evident. Scrub vegetation supported higher densities of adult aquatic insects dispersing further from the stream, in contrast to the rapid decline in open bedrock and forest. Furthermore, a comparison between downstream drift and aerial flight showed significantly more individuals where drifting downstream, and this method is liable to provide rapid recolonisation of macroinvertebrates within connected stream networks. At the local scale, organic matter (comprising leaves bags and timber) and artificial moss cover were added to six streams to determine if organic matter and habitat availability would improve macroinvertebrate communities in manipulated streams. A series of floods during the experiment reduced taxonomic richness and density in manipulated streams. Overall, this study has shown that after AMD has been treated, the geographic position of streams within the landscape and lateral dispersal barriers may prevent streams being rapidly re-colonised. Therefore, to promote rapid re-colonisation of macroinvertebrates, stream remediation projects should be targeted at streams that either have un-impacted headwaters or tributaries. This will allow macroinvertebrates to drift in and re-colonise faster. In my experiment I did not find that organic matter significantly enhanced the macroinvertebrate community, but moss additions did provide additional habitat for macroinvertebrates. These results highlight the importance that disturbance events can have on remediation projects.

Chapter One

Literature Review

Background

With unprecedented population and economic growth, humans are placing greater pressure on the world's water resources (Allan and Castillo 2007). Mineral resources, particularly coal, are coming under increasing pressure to support this growth (MED 2007). Meanwhile, advances in technology have made opencast coal mining particularly efficient and cost effective. However, opencast mining operations create water and sediment discharges that can have long-term adverse effects on freshwater environments (Gray 1998). These impacts on freshwater ecosystems are magnified by the transport of toxicants many kilometres downstream from a mine site. This contrasts with the localised effects of opencast mining on terrestrial communities (Gray 1998). The most extreme effects of open cast mining operations can be attributed to the generation of acid mine drainage (AMD) (Kelly 1988).

The extraction of coal began in earnest in New Zealand in the 1860s and was important for New Zealand (in terms of energy resources, regional wealth and employment) until the end of World War II. However, after the end of World War II the importance of coal diminished as hydroelectric power replaced coal fired power stations and diesel replaced steam trains (Henley 2007). In the late 1970s coal mining boomed again because of Japanese demand for high grade coal for their steel industries. Ever since, coal has remained a valuable source of regional employment and wealth. In the 2007/2008 financial year, New Zealand coal mines excavated 4.83 million tonnes of coal worth $600 million from the main coal mining regions of the Waikato, Buller, Greymouth and Southland (MED 2008). With increasing national and international energy security issues, New Zealand's coal resources are coming under greater pressure for exploitation. It is estimated that there are 15 billion tonnes of coal distributed across 35 coal fields. Otago and Southland coal fields are the largest with an estimated 7 billion tonnes of recoverable coal (MED 2009). The New Zealand government is actively encouraging exploration and exploitation of these resources to facilitate economic development and secure energy supplies. Because of the adverse effects coal mining can have on the environment (particularly AMD) it is necessary to understand the causes of degradation and improve our understanding of stream remediation. Ultimately, this will improve the environmental management of New Zealand's mining industry.

In this chapter I review our understanding of the effects of AMD on macroinvertebrate communities from the literature and then discuss the current issues that are important to the successful remediation of streams impacted by AMD.

Acid mine drainage and its longevity

Acid rock drainage (ARD) is a naturally occurring process when rock that contains elevated concentrations of sulphides is exposed to oxygen and water. Exposure of this material results in the oxidisation of sulphides, which generates sulphuric acid. However, when there is an increase in the surface area of rock during mining more sulphides (typically pyrite) are exposed to oxygen and air (Kelly 1988). Consequently there is an increase in the quantity of pyritic sulphur available for oxidisation and the generation of increased quantities of sulphuric acid. This is better known as acid mine drainage (AMD). The Buller region on the West Coast of the South Island is one of New Zealand's most severely AMD affected areas. The abundant Brunner Coal Measures (BCM) that dominates the area is particularly notorious for their AMD generation potential due to their estuarine development. This has led to pyrite and total sulphur content in the BCM ranging from 1.72 to 5.6 wt. % S (Nathan 1978).

AMD frequently occurs in catchments where the buffering capacity of the water is low. High concentrations of calcareous material in a catchment can neutralise mining derived acid. In addition, heavy metals in the surrounding rock are often released under low pH conditions, increasing concentrations of heavy metals to potentially toxic levels (Weber et al. 2006). If left untreated, AMD can persist for hundreds of years until sources of pyritic sulphur are exhausted (Gandy et al. 2009). AMD can have significant adverse effects on freshwater communities ranging from direct lethal effects from metal toxicity through to interference with ecosystem processes (Harding and Boothroyd 2004).

Effects of acid mine drainage on aquatic macroinvertebrate communities

The effects of pH

Aquatic macroinvertebrates tolerate a range of physico-chemical conditions present in their natural environment (Williams 1996). When exposed to conditions outside their tolerance ranges, individuals and populations can experience severe stress. In extreme cases, this can lead to their loss from the stressed system (Gray 1998). When streams that have poor chemical buffering capacities receive AMD, pH can be reduced to the tolerance levels of some species. Severe reductions in pH have resulted in the elimination of sensitive species, reduced taxonomic richness, and/or reduced macroinvertebrate abundance (Courtney and Clements 1998, Malmqvist and Hoffsten 1999, Winterbourn et al. 2000a, Gerhardt et al. 2005, Lin et al. 2005, Macedo-Sousa et al. 2007, Pond et al. 2008). For example, Van Damme et al. (2008) assessed the impacts of AMD on macroinvertebrate communities draining the High Andes in Bolivia and observed a significant decline in the number of macroinvertebrate families along an AMD impact gradient which ranged from pH 7.0-8.5 at reference sites to less than 3 at polluted sites. Similarly Feldman and Connor (1992) observed a reduction in macroinvertebrate abundance for 14 macroinvertebrate families and the elimination of 13 of 47 families in Virginia. Comparable patterns of lowered pH and a reduction in macroinvertebrate diversity have been found by Winterbourn and McDiffett (1996), Barry et al. (2000), Geurold et al. (2000), Berezina (2001), Lin et al. (2007), MacCausland and McTammany (2007). Typically, EPT taxa, especially mayflies are lost from the macroinvertebrate community while dipterans such as chironomids attain increased dominance (Winterbourn and McDiffett 1996, Gray 1998, Geurold et al. 2000, Lin et al. 2005).

However, not all macroinvertebrates respond in the same manner. For example, individuals that resided on the tops of substrate particles were eliminated faster than those deeper in the substrate, suggesting differences in exposure rates (Allard and Moreau 1987). Allard and Moreau (1987) also noted that sensitivity to pH seemed to change with different life stages. Thus, larger individuals of the mayfly *Ephemerella* seemed to be unaffected by reduced pH whereas early instar larvae died. Other studies have noted similar effects of lowered pH on macroinvertebrates at different life stages. Bell (1971) showed that at pH 4 Trichoptera adults (*Brachycentrus americanus* and *Hydropsyche betteni*) failed to emerge 50% of the time, whereas at pH 5.9 adult Ephemeroptera (*Ephemerella subvaria*) failed to emerge 50% of the time. Gerhardt (1990) found that *Baetis rhodani* only emerged 33 % of the time at pH 5 compared to pH 7. These results indicate reduced pH could potentially limit recruitment for following generations.

Changes in macroinvertebrate diversity and abundance do not always result from the death of individuals. Courtney and Clements (1998) showed that macroinvertebrate drift was a major mechanism for changes in abundance in a series of mesocosm experiments. However, variable responses between taxa to reduced pH were detected. For example, the caddisfly *Lepidostoma* sp. was significantly reduced in abundance, yet the net-spinning caddis *Arctopsyche grandis* showed no measureable response. Declines in taxonomic diversity and density may be driven by a variety of factors and were summarised by Hall et al. (1980) and Sutcliffe and Hildrew (1989). Firstly, pH can directly affect the physiology of organisms. Such an effect was observed by Frick and Herrmann (1990a) who found that at pH 4, the loss of sodium through osmoregulation was more pronounced than at pH 6. Secondly, pH is indirectly influenced by the effects of increased metal concentrations. Finally, changes to invertebrate communities may be driven by indirect effects of reduced primary production or decomposition, which impacts food availability to macroinvertebrates (Hall et al. 1980).

Low pH does not always have negative impacts on the invertebrate fauna, especially in areas where naturally acidic streams occur (Collier et al. 1990, Petrin et al. 2008). For example, Petrin et al. (2007a) compared streams in the north of Sweden that have naturally low pH to streams in the south which were experiencing anthropogenic acidification. Invertebrate communities in southern Sweden exhibited reductions in density and richness, particularly of mayflies. On the other hand, the northern streams contained diverse communities with numerous EPT taxa, despite pH as low as 4. Petrin et al. (2008) suggested macroinvertebrates in the northern Sweden streams were adapted to lower pH values. Similar results have been found in New Zealand brown water streams on the West Coast of the South Island, many of which are naturally acidic (Collier et al. 1990). Winterbourn and Collier (1987) observed that macroinvertebrate communities in acidic waters were comparable to those in circum-neutral streams until pH fell below 4.5. EPT taxa were represented in similar proportions in streams with pH ranging from 4.5 to a little over 7. Similarly, Collier et al. (1990) reviewed data from 43 naturally acidic streams on the West Coast of the South Island and found that taxonomic richness was not significantly correlated with pH. In fact, of the 37 most widespread taxa across all sampling sites (including circum-neutral streams) 92 % were found at sites with pH < 5.0 and 65 % were found at pH < 4.5. Their finding contrasts with the minimum threshold of 5.4-5.7, below which negative impacts of acidity were apparent in northern hemisphere studies which were reviewed by Sutcliffe and Hildrew (1989). Collier et al. (1990) suggested that in naturally acidic brown water streams in New Zealand dissolved aluminium was rendered non-toxic through a complex reaction with dissolved organic matter (DOC) (Collier and Winterbourn 1987). An alternative explanation is that species exposed to naturally acidic systems have evolved to withstand lower pH values (Graesser 1988, Weis and Weis 1989, Petrin et al. 2007a). This latter hypothesis was supported by the results of toxicity experiments with the common New Zealand mayfly, *Deleatidium*, which were more tolerant of pH as low as 3.5 if they came from streams with naturally low pH than from circum-neutral streams (O'Halloran et al. 2008). Additionally, the different responses of macroinvertebrates in anthropogenically acidified and naturally acidic streams could be influenced by metals, metal precipitates and their interaction with pH (Merrett et al. 1991).

Metal impacts

Determining the influence of low pH on aquatic macroinvertebrates is almost always compounded by the effects of dissolved metals, whose concentration is usually correlated negatively with pH (Merrett et al. 1991, Winterbourn et al. 2000a). The negative effect of elevated metal concentrations on invertebrate and fish communities has been well documented, with changes being characterised by reductions in diversity and density (Beltman et al. 1999, Hartmann et al. 2005, Van Damme et al. 2008, Iwasaki et al. 2009b). The greatest changes in community composition usually occur through the elimination of sensitive taxa such as Ephemeroptera, Plecoptera and Trichoptera (EPT taxa), and the dominance of tolerant taxa, such as chironomids. The ability of some chironomid taxa to maintain successful populations may in part be due to a reduction in the intensity of biotic interactions, particularly predation and competition (Hickey and Golding 2002). In Colorado streams polluted with cadmium, copper and zinc, invertebrate assemblages dominated by mayflies changed to communities dominated by Orthocladiinae (chironomidae) and tolerant caddisflies (Clements 1994). In microcosms, Kiffney and Clements (1994) found that mayflies collected from the Arkansas River in Colorado were most sensitive to metal pollution and were reduced in density by 65-90%, followed by stoneflies reduced by 36-60%, and caddisflies by 17%. More specifically, densities of the mayflies *Baetis tricaudatus*, *Epeorus longimanus*, *Rhithrogena hageni*, *Drunella grandis* and *D. doddsi* were reduced at concentrations of 1.1 µg/L of cadmium, 12 µg/ L of copper and 110 µg/L of zinc. Richardson and Kiffney (2000) found mayfly abundance was reduced by 76% and stonefly abundance by 50% when exposed to copper, zinc, manganese and lead in a ratio of 1:2.14:8.47:0.056, mg/L.

However, metals alone do not always affect stream communities negatively. In urban streams with high lead, copper and zinc concentrations, viable snail populations can be present (Blakely and Harding 2005). Winterbourn et al. (2000a) found that the common mayfly *Deleatidium*, the stoneflies *Spaniocerca* and *Spaniocercoides philpotti* and the predatory caddisfly *Psilochorema* were able to tolerate moderate metal concentrations in streams on the West Coast of the South Island, New Zealand. Moreover, Winterbourn (1998) found the caddisflies *Pseudoeconesus* sp. and *Kokiria miharo* in a stream with an aluminium concentration of 25.7 mg/L and an iron concentration of 8.2 mg/L.

Susceptibility of individual species to metal pollution has been shown to vary at different life stages. Juvenile *Potamopyrgus antipodurum* have been shown to be sensitive to copper concentrations of 54 µg/ L, whereas adults were are to tolerate concentrations of 79 µg/ L (Watton and Hawkes 1984). Similarly, Clark and Clements (2006) showed significant mortality of *Rhithrogena hageni* at sites contaminated with zinc, copper and cadmium during the summer when larvae were early instars. No mortality was observed in winter when organisms were larger. This relative sensitivity or tolerance of aquatic taxa to metals could be driven by exposure to elevated metal concentrations over evolutionary time (Harding 2005). Clements (1999) observed that *Baetis* and *Rhithrogena* mayflies from unpolluted streams were more sensitive to metal pollution than individuals from a stream with naturally elevated metal concentrations.

Reductions in diversity and density caused by metal pollution can be through 1) direct toxicological effects with mortality of individuals, or the avoidance of polluted water (McMurtry 1984), 2) physiological stresses (Herrmann and Andersson 1986) or 3) indirect effects such as the removal of top predators (Schultheis et al. 1997, Schorr and Backer 2006), or effects on food supply (Schlief 2004, Barnden and Harding 2005, Bray et al. 2008). Clements (1999) found drift rates of *Rhithrogena hageni* were four times higher when moved from a pristine stream to a stream with metal polluted (cadmium, copper and zinc) water flowing into it from an abandoned underground mine. Oligochaetes have also been shown to avoid metal contaminated sediment (McMurtry 1984). Physiological effects differ in that some organisms experience immediate mortality, whereas others have their fitness impaired. When concentrations of iron were increased 0 mg/L to 50 mg/L *Leptophlebia marginata* developed constipated midguts which ultimately led to death, probably by starvation (Gerhardt 1992). Subsequent dissection showed a thick layer of iron-humus-hydroxides built up on the midgut membrane (Gerhardt 1992). Moulting nymphs of *Ephemera danica* and *Heptagenia sulphurea* showed increased sodium loss indicating greater osmoregulatory stress when aluminium concentrations were elevated. As moulting is an energy demanding process, increased osmoregulatory stress can cause a trade-off growth/survival and emergence. A trade off between growth/survival and emergence could potentially limit recruitment as individuals may not have enough energy to emerge (Frick and Herrmann 1990b). Increased metal concentrations can also have positive effects. For example, aluminium can act as a barrier to sodium loss at low pH by working as an obstruction between the organism and the acidic water. The result is metals act as a barrier between the organism and acidic water preventing or reducing the negative consequences on osmoregulation (Frick and Herrmann 1990a). The presence of metals may also bring about changes in community composition or density indirectly if predatory species are eliminated and prey are released from predation (Soucek et al. 2002). The removal of intolerant taxa may also result in increased availability of food (e.g. algae) for more tolerant taxa (Ledger and Hildrew 2005).

Effects of sediment and iron precipitate

When appropriate management strategies are not in place, large quantities of sediment can enter waterways from opencast mines. These excessive sediment loads can have significant negative effects on macroinvertebrate communities (Ryan 1991). The impacts of sediment on macroinvertebrate communities include: the smothering of habitat (Suren et al. 2005, Matthaei et al. 2006, Townsend et al. 2008, Larsen et al. 2009), which has consequences for avoiding predation (McGrath et al. 2007), hydraulic forces (Yamada and Nakamura 2002) and disconnecting the stream bed from the hyporheic zone (Scarsbrook and Halliday 2002); reducing the quantity and quality of food resources such as periphyton); and increasing drift propensity (Shaw and Richardson 2001). Specifically, when sediment is deposited on the streambed it reduces habitat heterogeneity by smothering substrata and clogging interstitial spaces (Mol and Ouboter 2004). In six North Island streams impacted by gold mining, sedimentation was attributed to the decline in macroinvertebrate density (Quinn et al. 1992). In recently cleared tropical forest, artificial substrate was placed upstream and downstream of a sediment source (Watzen 2006). Downstream substrates experienced substantially more sedimentation and had significantly reduced EPT taxa and densities. In particular, taxa that were characterised as predators, scrapers and grazers were absent from substrates impacted by sediment (Watzen 2006). Experimentally adding fine and coarse sand to a stream in Brazil resulted in a significant reduction in invertebrate density and community richness in relation to a control. However, there was no significant effect of fine sediment on invertebrate communities as it was more easily exported from the stream during flooding (de Castro Vasconcelos and Melo 2008). However, drift rates of *Oxyethira albiceps*, *Paracalliope fluviatilis*, *Hydrobiosis* sp. and chironomid larvae were twice as high in experimental channels that had sediment added compared with controls (Suren and Jowett 2001), although, it could not be determined whether this was the result of smothered habitat, or reduced food (periphyton) supply (Suren and Jowett 2001).

The effect of sedimentation on macroinvertebrate food resources is typically to reduce photosynthesis of periphyton due to increased turbidity (Quinn et al. 1992, Parkhill and Golliver 2002, Molinos and Donohue 2009). Substrate available for colonisation by algae may also be more unstable due to sedimentation (Biggs and Kilroy 2004) and result in lower biomass. Autotrophic food resources affected in this way may limit food availability for consumers (Parkhill and Golliver 2002, Townsend et al. 2008) and have a cascading effect on the food chain (Wood and Armitage 1997). In the experimental channel experiment by Suren and Jowett (2001), chlorophyll *a* concentration was significantly reduced in the sediment treatment and Yamada and Nakamura (2002) also found a strong and significant relationship between suspended sediment and chlorophyll *a* concentrations. They concluded that reduced chlorophyll *a* concentration was the result of less light being able to penetrate the turbid water and consequently lowered photosynthesis. In contrast, elevated clay concentrations in experimental channels had no significant effect on the uptake of soluble reactive phosphorus in three, 90 minute treatments (Wolfe and Lind 2008). A larger exposure time may have resulted in an effect, however, as Molino and Donohue (2009) found that chlorophyll *a* was only reduced significantly after seven days exposure to increased clay concentrations. This finding suggests that chronic sedimentation events maybe more important than episodic events. Not only does sediment negatively impact periphyton, but it also has negative effects on macrophytes and bryophytes by reducing this photosynthetic efficiency (Reeves et al. 2004).The ultimate response of reduced habitat and food resources is for macroinvertebrates to leave the system (Shaw and Richardson 2001).

A sedimentation related issue is iron precipitate. When pH is lowered, metals are leached from the surrounding rock and are transported downstream in a soluble, dissolved form. However, as pH increases (the pH threshold metal precipitation is pH 3.5 to 4.3), iron precipitates from the water column and settles on the streambed in a similar way to other inorganic sediments (Harding and Boothroyd 2004, Barnden 2005, Harding 2005). Iron precipitate appears to be non-toxic to macroinvertebrates and periphyton (Dsa et al. 2008), but it can smother substrata, clog gills, and reduces visual acuity for macroinvertebrates (DeNicola and Stapleton 2002). Precipitates that smother periphyton communities result in reduced biomass (Wellnitz and Sheldon 1995, Verb and Vis 2000, Niyogi et al. 2002, de la Pena and Barreiro 2009).

Disruption of ecosystem function through effects on aquatic communities

Stream-dwelling organisms play vital roles in ecosystem processes such as organic matter decomposition, energy transfer and nutrient cycling (Wallace and Webster 1996). Consequently, these functions are likely to be impaired by changes in microbial, invertebrate and fish communities resulting from mining stressors. Perhaps the most important ecosystem process disrupted by AMD is organic matter breakdown (Niyogi et al. 2001, 2002, Schlief 2004, Barnden and Harding 2005, Schlief and Mutz 2006, Ehrman et al. 2008) through impacts on microbial and macroinvertebrate communities. Bacteria and fungi are responsible for conditioning of leaves and start the decomposition process (Webster and Benfield 1986). This microbial decomposition improves leaf palatability for detritivorous macroinvertebrates (‘shredders’), which consume leaves and generate fine particulate organic matter (FPOM) eaten by other macroinvertebrates, such as collector-browsers and filter feeders (Winterbourn 2004). However, at low pH and elevated metal concentrations, microbial activity is often impaired (Schlief 2004), and consequently the processing of leaves is also impaired (Schlief 2004, Barnden and Harding 2005, Schlief and Mutz 2006). Iron precipitates can also inhibit the processing of coarse particulate organic matter (CPOM) by smothering CPOM. The effect of both these processes is to reduce food resources throughout the food web (Niyogi et al. 2002, Schlief 2004, Barnden and Harding 2005, Schlief and Mutz 2006).

Remediation of acid mine drainage and the response of aquatic macroinvertebrate communities

If left untreated, AMD can persist for hundreds of years (Gandy et al. 2009). Due to increased environmental awareness and improvements in legislation over the last 20 years (e.g. the RMA 1991), new mining operations are expected to adequately treat or remediate AMD to reduce the effects on aquatic ecosystems. However, existing mines are coming under increasing pressure to remediate and reduce degraded mine discharges. Research into engineered systems that can treat AMD is relatively well advanced and globally, both active and passive treatment systems have been shown to work effectively to improve water quality (summarised by O'Sullivan 2005). In active treatment systems chemical reagents such as lime are added to the AMD discharge to neutralise acidic water and precipitate out metals into a sludge which is retained in a dam or sump. Active systems are usually employed on active mine sites, where electricity and maintenance staff are already on-site. In contrast, passive treatment systems, typically take the form of a wetland, and work though natural biogeochemical processes that reduce water acidity and metal loads (O'Sullivan 2005). Passive systems are usually low maintenance and are used at abandoned AMD sites. Despite these treatment systems being generally effective in reducing the discharge of pollutants to rivers, improvement in benthic invertebrate communities has been limited (e.g., Chadwick et al. 1986, Wantanabe et al. 2000, McClurg et al. 2007). This is because projects have tended to rely on the 'field of dreams' hypothesis (i.e., if we build it, they will come) where organisms are assumed to be able to re-colonise a restored system naturally (Palmer et al. 1997). In the case of AMD impacted streams, it is commonly assumed that if the stressor causing the degradation is removed, then the system will recover and re-colonise naturally. However, consideration needs to be given to the conditions that govern the construction of macroinvertebrate communities. A review by Lake et al. (2007) identified a number of important ecological components that need to be considered in remediation. In this thesis I have selected three of these components and investigated their importance in determining the recovery of macroinvertebrate communities. Below is a review of the importance of these components.

Factors limiting the successful recovery of aquatic macroinvertebrate communities

Sources of colonists

As many remediated AMD streams are inhabited by a few species they are usually reliant on random colonisation by macroinvertebrates from unimpacted source populations (Bond and Downes 2003). Many examples have been documented where mine drainage has been treated to improve water quality, yet invertebrate communities have failed to improve (Monteith et al. 2005, Armitage et al. 2007) or community recovery has taken longer than expected (Langford et al. 2009). Similar results have been found in channelised Scandinavian streams in forestry areas where stream morphology and habitat complexity has been restored, yet there has been limited recovery by the macroinvertebrate community (Muotka et al. 2002, Haapala et al. 2003). One reason for slow recovery of the macroinvertebrate communities may be a failure to recognise how regional constraints such as sources of colonists, and their ability to reach remediated reaches influence recolonisation rates (Palmer et al. 1997, Lowe 2002, Blakely and Harding 2005). The importance of sources of colonists for remediation was demonstrated in a wetland remediation project in Northland. To assess the recovery of a beetle community in rehabilitated mined peat bog a number of treatments were used to allow for rapid recovery. They included processed peat with no seed, processed peat with seed and direct habitat transplants. Results showed that beetle communities recovered fastest (towards a control beetle community) when habitats were transferred directly. This suggests, that when sources of colonists are located close by, recovery can be rapid (Watts et al. 2008).

Methods of macroinvertebrate re-colonisation

Five main pathways for re-colonisation of streams are possible, 1) via aerial dispersal of aquatic adults during their terrestrial life stage; 2) crawling and swimming upstream; 3) downstream drift; 4) emergence from the hyporheic zone; 5) transport of aquatic adults on other organisms (e.g., snails carried in the guts of birds) (Williams and Hynes 1976, Bilton et al. 2001). Many aquatic insects have an adult terrestrial phase where individuals can disperse large distances to colonise new habitats (Wallace et al. 1991, Bunn and Hughes 1997, Hughes et al. 1998, Briers et al. 2004, Chaput-Bardy et al. 2008). Aerial dispersal can be particularly important in river systems given their fragmented nature (Malmqvist 2002). An important issue in stream ecology has been how streams continue to maintain abundant populations in headwater reaches, which are liable to lose organisms due to downstream drift. To resolve this issue Müller (1954, 1982) proposed that downstream drift by aquatic macroinvertebrate larvae was compensated by upstream flight and oviposition of aquatic adults (the colonisation cycle). Numerous studies have addressed this hypothesis and investigated flight. Upstream flight has been found to predominate in some species but not in others (Peckarsky et al. 2000, Winterbourn and Crowe 2001, Collier and Quinn 2003, Elliott 2003). For example, in the Kuparuk River in Alaska Hershey et al. (1993) isotopically enriched macroinvertebrate fauna with ^{15}N and found 33-50 % of macroinvertebrates were drifting about 2.1 km downstream. This was compensated by one third to half of all emergent adults flying 1.6-1.9 km upstream. Elliott (2003) found that in nine out of 10 species in the United Kingdom upstream flight dominated; predators such as *Perlodes*, *Rhyacophila* and *Isoperla* being particularly strong fliers that migrated upstream at a rate of 9.5- 13.5 m per day. In a New Zealand forest stream, caddisflies such as *Zelandopsyche ingens* and *Hydrobiosella stenocerca* were significantly more abundant on the downstream side of sticky traps, suggesting up-stream flight (Winterbourn and Crowe 2001). Fraser (2006) also found that caddisflies were predominantly up-stream dispersers whereas mayflies and stoneflies exhibited no directional preference.

Contradictory to Müller's (1954, 1982) colonisation cycle theory, Anholt (1995) argued that up-stream flight was not enough to maintain macroinvertebrate populations. Rather density dependence of communities helps to maintain populations as well as random dispersal events from other streams. These random dispersal events commonly occur via lateral dispersal of adult aquatic insects. It is a potentially important mechanism for the colonisation of adjacent disconnected streams (Bunn and Hughes 1997, Wilcock et al. 2003, Briers et al. 2004, MacNeale et al. 2005a, Zickovich and Bohonak 2007, Chaput-Bardy et al. 2008). Some studies have indicated widespread lateral dispersal (Jackson and Resh 1989, Miller et al. 2002, Briers et al. 2004, MacNeale et al. 2005a, Wilcock et al. 2007, Zickovich and Bohonak 2007), whereas in others it has been minimal or non existent (Collier and Smith 1998, Winterbourn 2005, Finn et al. 2006, Finn et al. 2007). In one of the earliest studies to investigate lateral dispersal, Jackson and Resh (1989) found that the total number of individuals and the total biomass exponentially decreased with increasing distance from the source stream (Figure 1.1). They concluded that the distances species were able to disperse were related to the individual species flying ability. Winterbourn (2005) found that 98 % of all stoneflies were caught along the stream edge and the species Spaniocerca zelandica and Cristaperla fimbria were the most abundant. The limited dispersal distances of stoneflies were attributed to their poor dispersal abilities as reported by Hughes (1999), and Briers et al. (2003, 2004). In contrast, Griffith et al. (1998) found one strong flying caddisfly species (*Pseudostenophylax uniformis*) in a West Virginian stream that showed no reduction in numbers with increasing distance from the stream.

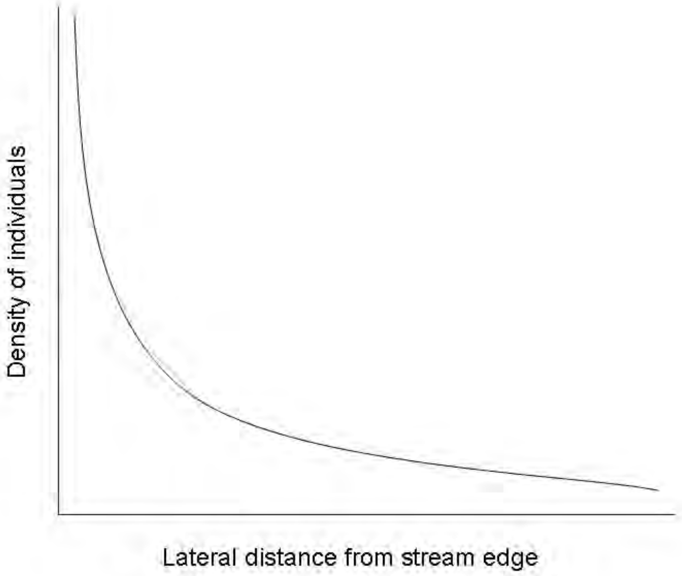


Figure 1.1: The typical relationship between invertebrate density and dispersal distance from stream edges (modified from data from Jackson and Resh (1989), Petersen et al. (1999) and Winterbourn (2005)).

An alternative to direct trapping methods such as Malaise or sticky trapping is to use molecular or nutrient enrichment techniques to determine existence and dominance of lateral dispersal (e.g. Briers et al. 2004, MacNeale et al. 2005a, Finn et al. 2007, Zickovich and Bohonak 2007). Using molecular techniques a number of authors have found very little genetic differentiation of macroinvertebrates between streams across different catchments up to 20 km apart, suggesting widespread lateral dispersal (e.g. Miller et al. 2002, Wilcock et al. 2003, Wilcock et al. 2007, Zickovich and Bohonak 2007, Chaput-Bardy et al. 2008). Nevertheless, these findings must be viewed with caution as mass dispersal over large spatial and temporal scales might not be driving these patterns. Rather, migrations of only a few individuals may overtime produce self sustaining populations that are genetically similar to those in distant streams. This phenomenon was been observed in two nutrient enrichment field investigations. The first was by Briers et al. (2004) where 1.5 million *Leuctra inermis* were labelled with ^{15}N. Surveys estimated that 0.1-0.2 % of individuals were dispersing about 1 km from their natal stream. Similarly, a small numbers of female stoneflies (*Leuctra ferruginea*) that were labelled with ^{15}N were found at a non-enriched tributary 500 m away (MacNeale et al. 2005b). Bunn and Hughes (1997) estimated that only 3-12 individuals were needed to generate larval densities of between 2 and 6 m^2 in an Australian caddisfly in small headwater streams. Thus, a self-sustaining population could develop in one or two generations. This could explain why a number of studies have found genetic similarity between macroinvertebrate populations in different catchments (Milner et al. 2000, Schultheis and Hughes 2005, Wilcock et al. 2007, Zickovich and Bohonak 2007, Chaput-Bardy et al. 2008) up to 20 km apart (Bunn and Hughes, 1997), despite adults being largely restricted to the immediate vicinity of the natal stream itself (Collier and Smith 1995, Petersen et al. 1999).

Patterns of genetic differentiation that are used to infer connectivity of populations can be site specific and dependent on a number of factors such as topography and vegetation. Mountain ranges can act as dispersal barriers (Wilcock et al. 2003, Finn et al. 2006, Finn et al. 2007, Lehrian et al. 2009) and vegetation has been shown to influence on lateral dispersal distances. Thus, Delettre and Morvan (2000) found that dense vegetation stands hindered the lateral dispersal of chironomids in contrast to open land. Similarly, Collier and Smith (1998) found that lateral adult flight distances were restricted within riparian forest, possibly because forest stands were too dense to allow adults to successfully navigate. They observed for females that the decline of individuals with increasing lateral distance increased had two peal densities. For both *Olingia feredayl* and *Pynocentrodes aureola* this was right by the stream edge. But a second peak was recorded 50 m from the stream edge for both species. This was postulated to be the result of females resting on the vegetation allowing males to find them for mating and allowing eggs to mature (Collier and Smith 1998).

Downstream drift is another important dispersal mechanism. It can be induced by a variety of abiotic and biotic factors such as increased turbidity (Sweka and Hartman 2001), pollution (Clements 1999, Cole et al. 2001, Heckman and Friberg 2005), floods (Bond and Downes 2003, Mochizuki et al. 2006, Gibbins et al. 2007), drought (reviewed by (Dewsen et al. 2007), density-dependent interactions (Turner and Williams 2000, Shearer et al. 2003), or accidentally (Allan and Castillo 2007). Drift can be an especially important source of colonists for remediated reaches when there are healthy headwaters or tributaries (MacCausland and McTammany 2007). Drift can also provide an immediate supply of colonists compared to aerial dispersal (Kramer 1982), which is often seasonal and subject to the incubation and development of eggs (MacKay 1992). While the importance of drift as a source of colonists was recognised by Müller (1954), it was first quantified by Williams and Hynes (1976) and Townsend and Hildrew (1976). Specifically, Townsend and Hildrew (1976) found that downstream drift accounted for 82 % of macroinvertebrates colonising disturbed substrate. In the south branch of the Waimakariri River in the South Island, Kramer (1982) considered the pathways of colonists to disturbed substrate. From this study it was concluded that drift accounted for 43 % of colonists while adult flight only accounted for 13 %. Numerous subsequent studies investigating the importance of drift have focussed on colonisation of systems disturbed by flooding (Zbinden et al. 2008). Matthaei et al. (1997) found that trays suspended in the water column developed a similar community composition to trays that were placed on the streambed implying that drift was the most important source of colonists. Similar conclusions have been reached by numerous other workers (Paltridge et al. 1997, Fenoglio et al. 2002, Death 2004, Duran 2006)

Re-colonisation via drift is likely to be dependent on the species composition of the stream. For example, Matthaei et al. (1996) found that the only taxon that was abundant in drift samples was Simuliidae. In contrast, re-colonisation by the mayflies *Baetis* and *Rhithrogena* was probably accomplished by swimming and crawling from surrounding substrate. The importance of drift as a re-colonising mechanism is also determined by the position of the reach within the river continuum. In mine impacted systems, it is possible that streams will have their headwaters destroyed and even after the catchment has been rehabilitated and AMD treated, up-stream sources of macroinvertebrates will not be present. Therefore, macroinvertebrates must re-colonise primarily by up-stream flight (Wallace et al. 1986).

The hyporheic zone is another potential source of colonists. This is the area of interstitial space beneath the stream bed that is saturated with stream water (White 1993) and has been recognised as a source of colonists after natural disturbance events such as flooding and drought (Cooling and Boulton 1993, Matthaei et al. 1999, Bo et al. 2006). In North Island streams disturbed by flooding, Fowler (2002) found that the hyporheic zone was less important than drift as a source of colonists. Furthermore, different species colonised from different sources. For example, *Deleatidium* colonised from the hyporheic whereas elmid beetles were primarily found in drift samples. While most studies have found that the hyporheic zone does supply colonists, it is only of secondary importance when compared to drift (Matthaei et al. 1999, Fowler 2002). Because recolonisation from the hyporheic zone can only involve taxa present within an affected stream reach, it is unlikely to provide colonists in streams affected by mining where heavy metals and low pH have eliminated the sensitive taxa (Anthony 1999). Furthermore, iron precipitate can cement substrate particles together in acidified reaches disconnecting the hyporheic zone from upper streambed (Anthony 1999, Barnden 2005).

Habitat availability

Habitat complexity influences the composition of macroinvertebrate communities on a local scale (Downes et al. 2000, Johnson et al. 2003, Heatherly et al. 2007, Schneider and Winemiller 2008b). Habitat complexity is created by organic matter such as macrophytes, bryophytes, coarse woody debris (CWD) and organic matter debris jams, as well as inorganic matter such as boulders and cobbles (Suren 1991, Death 2000, Milner and Gloyne-Philips 2005, Heino and Korsu 2008). This habitat provides a variety of refuge for stream dwelling organism to exploit to protect themselves from flooding and predation as well as providing food resources (Allan and Castillo 2007) During open cast mining, much of this habitat can be degraded through sedimentation and loss of organic matter sources (Harding and Boothroyd 2004). When remediating AMD-impacted streams the focus is usually on improving water quality discharges and typically does not focus on improving in-stream habitat. Consequently, limited habitat complexity and availability can impact on the recovery of macroinvertebrate communities (Suren et al. 2004). Therefore, when remediating AMD impacted systems it is important to consider in-stream habitat improvements to facilitate macroinvertebrate recovery. Manipulative studies have attempted to alter substrate composition in remediated stream reaches, and these have produced mixed results. For example, Sarriquet et al. (2007) added cobble substrate to a restored reach of the Tamoute River in Northern Brittany, France and observed no increase in invertebrate density or taxonomic richness. However, community composition shifted towards being dominated by species with an affinity for coarser sediments. These species included the mollusc *Ancylus fluviatilis*, Trichoptera (*Wormaldia* sp. and *Plectrocnemia* sp.), Ephemeroptera (*Baetis* sp. and *Habroleptoides* sp.) and

Plecoptera (*Amphinemura* sp.). The absence of a macroinvertebrate density response was attributed to a major drought during the study period (Sarriquet et al. 2007). The remediation of channelised streams in Finland by inserting boulders into the stream had no significant influence on the macroinvertebrate communities (Tikkanen et al. 1994) possibly because boulders typically have lower macroinvertebrate richness and density than cobble substrate (Quinn and Hickey 1990). While the evidence suggests boulders may not provide useful macroinvertebrate habitat they can be important for oviposition and can create habitat and flow variability in this vicinity (Reich and Downes 2003). In an urban stream restoration project in Christchurch, it has been found that after extensive riparian vegetation planting and in-stream habitat additions had occurred there was no significant improvement in the macroinvertebrate community. Reasons that were identified for this lack of macroinvertebrate response was because of a lack of desirable source populations of macroinvertebrates, sedimentation and undesirable stream flows (Winterbourn et al. 2007).

To my knowledge no published studies have investigated the re-introduction of moss to streams that have undergone remediation. However, the importance of moss as a habitat has been well documented (Maurer and Brusven 1983, Brusven et al. 1990, Suren 1990, Suren 1991, Suren and Duncan 1999, Hutchens et al. 2004, Heino and Korsu 2008). For example, moss grown in experimental channels in Idaho, had 5-30 times the insect density and twice the macroinvertebrate biomass than the surrounding mineral substrate (Maurer and Brusven 1983). The elevated macroinvertebrate density in moss may have been a response to an increased food resource of trapped organic matter. In a subsequent study in which moss cover was manipulated in experimental channels, channels with added moss (20 % cover) had 2-7 times greater macroinvertebrate density and 1.5-6 times greater macroinvertebrate biomass than channels without moss cover (Brusven et al. 1990). The complete removal of moss from Appalachian bedrock streams resulted in a significant decline in macroinvertebrate diversity, abundance and biomass (Hutchens et al. 2004).

Tikkanen et al. (1994) used an excavator to place boulders and cobbles in a stream channel to enhance the retentive capacity of the channel. After an initial decline in macroinvertebrate density as a result of the manipulation, re-colonisation resulted in pre-disturbance densities being reached within one month. Furthermore, moss covered stone that were present prior to the excavator work supported significantly more macroinvertebrates than those without moss cover, suggesting that mosses can act as a refuge for macroinvertebrates during disturbances, and also may provide a source of recolonists (Korsu 2004). The studies by Tikkanen et al. (1994), Hutchens et al. (2004) and Korsu (2004) all indicate the importance of moss for providing habitat, refuge and a potential source of colonists. The re-introduction of moss and other bryophytes should be considered when remediating a stream reach from human disturbance.

In catchments affected by mining, quantities of coarse woody debris (CWD) can be reduced in quantity, especially after the removal of riparian vegetation. However, although I have not been able to identify any studies that have quantified the effect of reduced CWD on macroinvertebrate communities during mining its effects can be inferred from studies of other land-use impacts. For example, conversion of the riparian zone from native forest to pasture reduced CWD inputs by 80 % in New Zealand streams (Quinn et al. 1997). Recent remediation projects that have re-introduced CWD provide good evidence for its ecological importance (Bond et al. 2006). The ecological roles of CWD include providing: a) complex structural habitat complexity for macroinvertebrates and fish (Lemly and Hilderbrand 2000, Bonnett and Sykes 2002, Rowe and Smith 2003, Mossop and Bradford 2004, Lester et al. 2007, Lester and Boulton 2008); b) refuges from predation (Everett and Ruiz 1993); c) direct food resources (Collier and Halliday 2000); d) trapping of other detrital resources (Lemly and Hilderbrand 2000, Scealy et al. 2007); e) sites for oviposition (Dudley and Anderson 1982); and pupation (pers. ob.).

The ecological importance of leaves and wood provides a rationale for the re-introduction of CWD into remediated streams. Furthermore, the addition of CWD can be a particularly cost effective means of re-establishing complex habitat to a system, which may have experienced heavy sedimentation (Scealy et al. 2007). The addition of CWD may also promote more immediate macroinvertebrate community responses compared to other kinds of habitat additions and modifications. For example, in a remediated agricultural stream Lester et al. (2007) added equal volumes and surface areas of wood into 50 m reaches in eight streams. The response of the macroinvertebrate community was a significant increase in the abundance of families from each functional feeding group in the streams. More complex food webs were also detected as the mean number of predatory macroinvertebrate families in CWD treatment habitats was 3.86 compared to 1.66 in benthic inorganic substrate. Higher richness of shredder families was also observed in CWD habitat, and was most likely the result of increased deposition of CPOM (Lester et al. 2007). When habitat complexity was enhanced by the additions of CWD to streams it was found that mean abundances of macroinvertebrates were always greater on the more complex CWD than benthic substrate. This was attributed to increased availability of habitat and retrained organic matter compared to streambed substrate. Macroinvertebrate densities also increased, up by 26 % in the complex CWD compared to simple benthic habitat (Scealy et al. 2007a). In Okeover stream (an urban stream in Christchurch) it was found that wood and leaf packs did not support greater taxonomic richness or macroinvertebrate density than stony substrate (Cottam 1999). These results suggest that leaf packs and wood do not provide additional habitat for specialist taxa, rather they provide an additional habitat that can be utilised. However, wooden habitat can be important in streams that have experienced severe sedimentation, or are naturally fine sediment bottom streams. For example, additions of CWD to low gradient, sandy bottom streams in Georgia State supported 60 % of total macroinvertebrate production despite CWD only comprising 4 % of instream habitat (Benke et al. 1985). These CWD habitats were also important contributors to taxonomic richness in Michigan and Minnesota streams with the wood increasing mean total taxa by 55 % and 26 % respectively (Johnson et al. 2003).

Organic matter food resources

Organic matter from the riparian zone is an important food resource for many macroinvertebrates, especially in headwater streams (Wallace et al. 1997, Li and Dudgeon 2008, Doi 2009). As many headwater streams are heterotrophic, the loss of riparian vegetation during mining can have significant impacts on macroinvertebrate communities (Wipfli et al. 2007). The loss of habitat heterogeneity and retentive structures can decrease organic matter retention (Ehrman and Lamberti 1992, Muotka and Laasonen 2002). For example, in forestry catchments, particularly in Scandinavia, north-west Russia, America and Canada streams have been channelised to aid timber transport. The result has been the loss of boulders, over hanging vegetation, CWD and natural meanders that retain organic matter (Lepori et al. 2005). The loss of this retentive habitat resulted in organic matter being rapidly exported, limiting the food resources of macroinvertebrates (Lepori et al. 2005). Similar findings have been made by Hall et al. (2000), Baer et al. (2001), Eggert and Wallace (2003a, b) and Doi (2009).

In some remediation projects, practitioners have set about improving organic matter resources in streams so that detritivore communities can be re-established (Muotka and Laasonen 2002). The most common approach in channelised streams is to improve their retentive capacity (Lepori et al. 2005a). Haapala et al. (2003) released 1000 strips of plastic (8 x 4 cm), which replicated leaves, to identify leaf retentive zones. The results showed that densities of shredders and other detritivores were higher in retentive sites where there was significantly more organic matter trapped. As a result, retention structures should be focused in runs and riffles. Negishi and Richardson (2003) found that placing clusters of boulders in the substrate boulder increased organic matter storage by 550%, and resulted in an almost 300% increase in total invertebrate abundance, primarily detritivores. A similar result was found by Muotka and Syrjanen (2007).

Thesis structure

This thesis has been written as four independent manuscripts. Consequently, there is some repetition between the literature review and the introductions of each manuscript.

Chapter Two describes the results from a survey of 45 streams on the Stockton Plateau, north of Westport on the West Coast of the South Island. Specifically, the survey identified the location of high diversity streams which might act as sources of recolonisers and assessed the effect of environmental variables on macroinvertebrate community composition. Chapter Three is an investigation into the aerial flight directions of adult insects as well as considering the influence of vegetation on lateral dispersal distances. This chapter also considers the relative importance of both downstream drift and downstream aerial flight for supply re-colonists to remediated reaches. Chapter Four focuses on in-stream factors which might be used to enhance the remediation of benthic communities. Specifically, experimental manipulations were made by adding habitat and organic matter. Finally, a synthesis of my key findings from this research as well as a stream “remediation check list” is presented in Chapter Five.

Chapter Two

Environmental and spatial determinants of stream macroinvertebrate communities in an acid mine drainage impacted landscape.

Introduction

Acid mine drainage (AMD) is a common phenomenon affecting many mining regions around the world (Kelly 1988). This feature is a significant issue because it can affect entire river systems, can extend many kilometres downstream from a localised disturbance footprint (Gray 1998), and can persist for centuries (Gandy et al. 2009a). AMD is generated when pyritic sulphur is exposed during mining operations to both oxygen and water. The resulting oxidisation process produces sulphuric acid (Kelly 1988), which typically lowers stream pH and increases dissolved metal concentrations (Harding and Boothroyd 2004). Stream macroinvertebrate communities are impacted by AMD through direct and indirect physiological effects (Herrmann and Andersson 1986, Gerhardt 1990, Frick and Herrmann 1990a, Frick and Herrmann 1990b, Herrmann et al. 1993, Gerhardt et al. 2005, Brinkman and Johnston 2008) and disruption of ecosystem processes (Niyogi et al. 2001, Schlief 2004). As a result of these interactive effects, taxonomic richness is reduced and densities decline (Winterbourn and McDiffett 1996, Courtney and Clements 2000, Winterbourn et al. 2000a, Lin et al. 2005, MacCausland and McTammany 2007, Van Damme et al. 2008). To reduce these effects, many mining companies treat AMD before it is discharged from a mine area, enabling macroinvertebrate communities to recover naturally. However, several remediation projects have demonstrated that improved water quality does not always result in macroinvertebrate community recovery (Monteith et al. 2005, Armitage et al. 2007, Iwasaki et al. 2009, Langford et al. 2009). A consistent problem in many failed stream remediation projects is the lack of colonist sources (Lake et al. 2007, Langford et al. 2009). Failures are commonly caused by stream restoration practitioners not considering the importance of how macroinvertebrate communities are assembled.

In a landscape comprised of pristine and remediated streams, individual macroinvertebrate communities may form a meta-community. This meta-community consists of source systems (pristine streams) providing individuals to sink systems (remediated streams). There are three main concepts that are frequently used to explain how such meta-communities are structured. These are: 1) the patch dynamic paradigm, in which communities are structured by dispersal limitation and predator-prey interactions, 2) the species sorting paradigm, where meta-communities are limited by abiotic conditions, and 3) the mass-effects paradigm, which suggests sufficient dispersal exists to allow unsuitable habitats to be colonised (Leibold et al. 2004). Therefore, the macroinvertebrate community will be assembled by the strength of hierarchy between regional and local constraints stipulated by Palmer et al (1997). Regional constraints include the regional pool of species, the dispersal abilities of different species, and biogeographical factors. Local constraints consist of habitat availability and food resources (Palmer et al. 1997).

There are a number of studies that have demonstrated the importance of landscape position and environmental factors in structuring macroinvertebrate communities (e.g. Mykrä et al. 2007, Heino and Mykrä 2008, Ilmonen et al. 2009, Sheldon and Warren Jr 2009). However, few studies have considered how the effects of AMD on macroinvertebrate communities can impact the function of meta-communities, or how this is likely to influence the successful remediation of macroinvertebrate communities (but see Masters et al. 2007). Therefore, before any attempt at biologically remediating a stream treated for AMD, it is important to determine if macroinvertebrate communities are being structured by the spatial arrangement of streams or by local environmental conditions. It is also vital to identify streams that could potentially act as colonist sources. Consequently, the aim of this study was to determine the relative influence of the geographic position of streams within the landscape and environmental factors on macroinvertebrate richness and community composition in a landscape matrix impacted by AMD.

Method

Study Sites

The Stockton Plateau is 500-700 m asl, 21 km east-northeast of Westport on the West Coast of the South Island of New Zealand (Figure 2.1). This plateau supports the largest open cast coal mine in New Zealand, the Stockton Mine (Henley 2007). The Stockton Plateau is a characterised by a climatically extreme environment. The predominant wind direction is from the west and north-west. This wind is forced to ascend the 1,100 m steep western face of the plateau, where cooling produces orographic rainfall. Annual rainfall on the plateau is about 6,400 mm per year (Alarcon 1997). Coal in this region is located in the Mangatini seam of the Brunner Coal Measures (BCM). The BCM overlay a basement of sedimentary greywacke and argillite from the Ordovician period (480 million years ago). BCM comprise a sequence of coarse-grained quartzo-feldspathic sandstone and granule to cobble conglomerate. Stockton coal lays over this sequence, which is then overlaid by a layer of mudstone and fine to medium sandstone, which is up to 4 m thick. Plant communities on the plateau are products of the extreme climatic conditions. On the rock pavements, prostrate and upright manuka (*Leptospermum scoparium*) as well as grasses and mosses dominate. These stunted communities are the result of shallow, infertile, acidic (pH 3.5-5.6) and poorly drained soils (Alarcon 1997, Theinhardt 2003). The more gently sloped areas of the plateau are dominated by manuka (*Leptospermum scoparium*), carpet grass (*Chinochloa australis*), red tussock (*C. rubra*), wire rush (*Juncus* sp.) and *Chionochloa* species. Lowland forest communities are present in gullies and eastern facing slopes. These communities include; southern rata (*Metrosideros umbellata*), mountain beech (*Nothofagus solarnadri v. cliffortioides*), mountain Toatoa (*Phyllocladus alpinus*), yellow-silver pine (*Lepidothaminus intermedius*), pygmy pine (*Lepidothamnus intermedius*), *Dracophyllum densum* and *Dracophyllum politum* (Alarcon 1997, Theinhardt 2003). Above the tree line, vegetation is dominated by subalpine vegetation such as leatherwood (*Dracophyllum longifolium*), mountain flax (*Phornium cookianum*), *Carpha alpina*, snowberry (*Gaultheria depressa*), and mountain daisy (*Clemisa* sp.) (Theinhardt 2003).

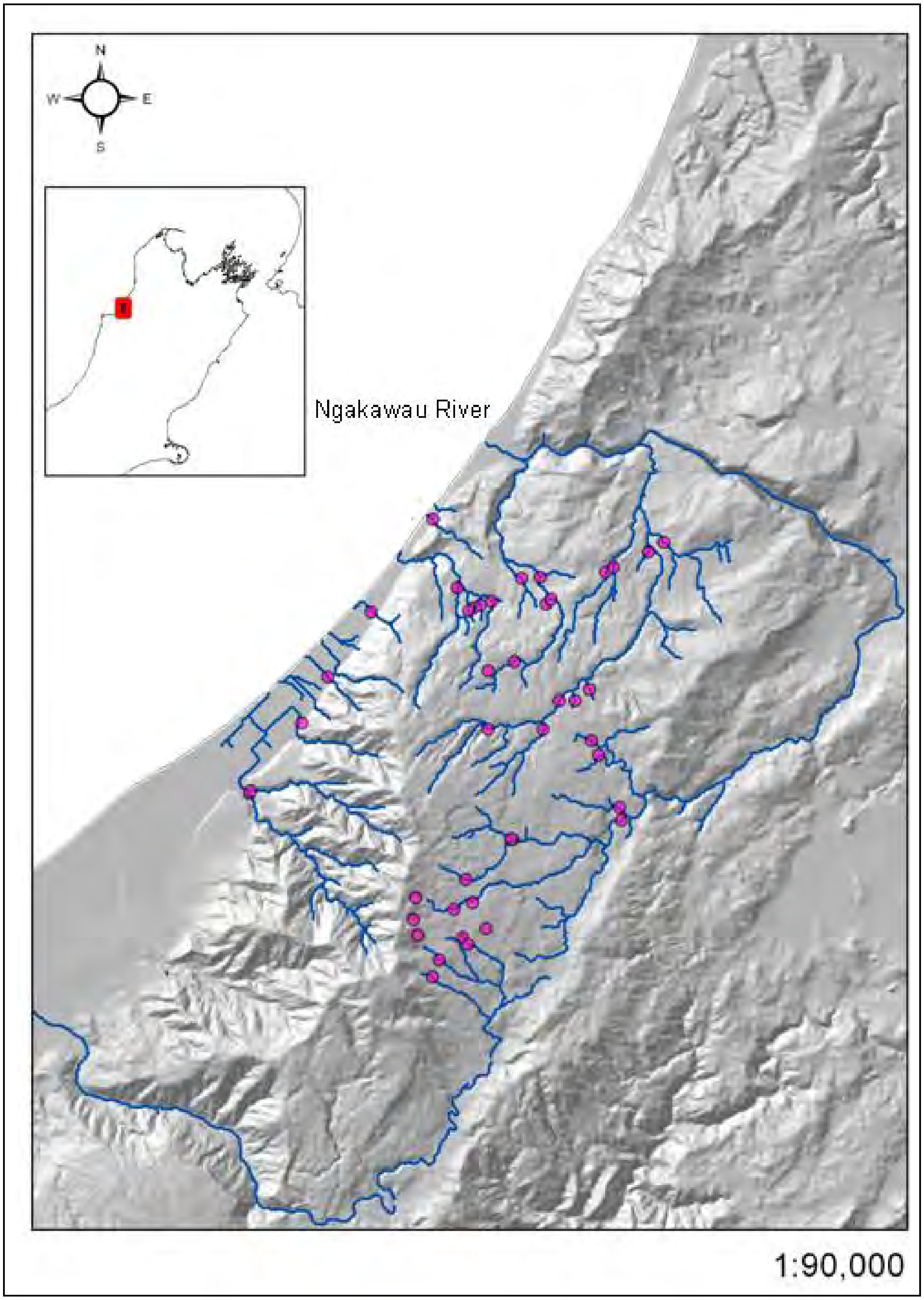


Figure 2.1: The Stockton Plateau, north-west of Westport, New Zealand showing all streams (ephemeral and perennial) draining the plateau. Of these, 45 streams (pink dots) had physico-chemical measurements and macroinvertebrate samples collected once between November 2007 and January 2008. Stream names and physico-chemical data is in Appendix 1.

Physico-chemical Sampling

A total of 45 streams were sampled across the Stockton Plateau on a single occasion during the austral summer of December 2007 to January 2008. Streams were selected from topographic maps and expert site knowledge so that a reprehensive subset of impact, mildly impacted and natural streams were sampled. At each site a range of physical and chemical parameters were measured (temperature, specific conductance and pH) using a Eutech meter (Oakton 10 series, Oakton Instruments, Vernon Hills, Illinois, USA). In addition, a single grab water sample was collected from each site for analysis of dissolved aluminium, iron, nickel and zinc. Samples were filtered through a Whatman 0.45 µm membrane filter into a 250 ml polyethylene bottle. All bottles were treated with nitric acid to retain metals in solution prior to being analysed. Dissolved metal samples were measured by Hills Laboratories (Hamilton, Waikato) using an Inductively Coupled Plasma-Mass Spectrometer (ICP-MS). Two additional water samples were collected in rinsed polyethylene bottles for the measurement of acidity, total suspended solids (TSS) and turbidity (NTU). TSS was measured in the laboratory using the methods outlined in APHA (2005) while turbidity was measured with a Hach 2100N meter.

At each site stream channel stability was assessed in a 50 m using the index developed by Pfankuch (1975). Where present, moss cover was estimated in a 0.04 m^2 grid. Physical characteristics of each stream were also measured. Wetted stream width was measured at five locations within a 50 m reach, and at each width, stream depth was measured at five equal intervals. Surface water velocity estimates were made by timing a float over a known distance three times in the middle of the stream. Substrate composition was recorded as well as the dominant riparian vegetation.

Macroinvertebrate sampling

Benthic macroinvertebrates were collected in a 50 m reach that included at least one riffle and one run. At each site, three Surber samples (0.04 m^2; 500 µm mesh) were collected within randomly selected riffles and runs. The substrate was vigorously disturbed to about 10 cm. In addition, one extensive kick-net (500 µm mesh) sample was collected from a range of microhabitats within the reach following the protocols of Stark et al.

(2001). Samples were preserved in the field with 70% ethanol and returned to the laboratory. In the laboratory, the samples were picked and macroinvertebrates enumerated. Identifications were made under a binocular microscope, (x 10) to the lowest taxonomic level possible using keys by Smith (2001) and Winterbourn et al. (2000b).

Statistical Analysis

Initially all sites were divided into three impact categories based on expert knowledge of the Stockton Plateau. The classes were: severely impacted (sites with direct AMD inputs); mildly impacted (sites that either had sedimentation issues associated with mine infrastructure or were naturally acidic), and; reference sites with a pH > 5. These categories were confirmed by a detrended correspondence analysis (DCA) in PC-Ord (McCune and Mefford 1999).

The effect of dissolved metals, pH and other environmental variables on macroinvertebrate community metrics were analysed using a combination of best subsets regression and ordination procedures. Each of the four metals (Fe, Al, Zn, Ni), and conductivity were highly correlated ($r > 0.88$). Thus, prior to analysis, the four metals and conductivity were reduced into a single variable ‘metal index’ with Principle Components Analysis (PCA). Data was log transformed prior to analysis in PC-Ord (McCune and Mefford 1999). The PCA produced a single, highly significant axis that explained 90 % of the variation in each of the five metal variables. This ‘metal index’ was used in all subsequent analysis that tested any relationship with metals.

Macroinvertebrate community metrics used included the Macroinvertebrate Community Index (MCI) (Stark 1985) and the Quantitative Macroinvertebrate Community Index (QMCI) (Stark 1993). These indices are based on the tolerances of macroinvertebrates in riffles of hard bottom streams to organic enrichment (Stark, 1985) and allow for classification of sites into categories: clean water; doubtful quality or possible pollution; probable moderate pollution; probable severe pollution (Stark 1998). Other metrics used included the proportion of Ephemeroptera, Plecoptera and Trichoptera (ETP) taxa, total richness, macroinvertebrate density, functional feeding groups (Predators, Shredders, Grazers, Collector-gathers, and Filter-feeders) (Cowie 1980), as well as percent chironomids and Molluscs. Community evenness was also calculated using the Berger-Parker index. This index expresses the proportional importance of the most dominant species (May 1975). Margalef's index, which is another measure of species diversity, was also calculated (Magurran 1988).

In order to test the relationships between macroinvertebrate community composition and environmental variables, we used multivariate ordination techniques in CANOCO (Leps and Smilauer 2003). For the macroinvertebrate community abundance data the three Surber samples were averaged. These data were then log-transformed to meet the assumptions of normality. Environmental variables were also log-transformed with the exception of pH. I then used redundancy analysis (RDA), a linear ordination technique, because an initial detrended correspondence analysis (DCA) indicated a short community gradient (2.9) (Leps and Smilauer 2003).

Significant explanatory environmental variables that explain macroinvertebrate community composition and abundance were selected using the hierarchical stepwise forward selection procedure in CANOCO. This procedure performs a series of intermediate partial ordinations. Initially all variables are fitted to the species matrix individually and tested for significance using 199 Monte Carlo permutation tests. The most significant ($p < 0.05$) variable is retained and used as a covariate for an analysis of all remaining variables. The next most significant variable is then retained and the process continues until all significant variables are found (Leps and Smilauer 2003).

To assess the relative importance of landscape position and environmental variables on macroinvertebrate community composition, a partial RDA (pRDA) was used. The purpose of this pRDA was to partial out the effects that landscape position and local environmental variables independently and jointly have on structuring macroinvertebrate communities. Easting and northing co-ordinates were used to identify the landscape position of the sampling sites. The different components of variation derived from the analysis were (E) spatially structured environmental variation, (S) environmentally structured landscape position variation, (E-S) pure environmental and (S-E) pure landscape position. The significance of each component was determined using 199 Monte Carlo permutations. Using these results, it was then possible to calculate explained variation [(E) + (E-S)], the unexplained variation [1-(E) + (E-S)] and the shared variation between (E), (S) and (E)- (E-S) (Cottenie 2005).

Best subsets regression was conducted for all univariate community indices against environmental variables using STATISTICA (StatSoft 2008). Best subsets regression is used to determine which predictor (independent) variables should be included in a multiple regression model. It operates by examining all of the models created from all possible combination of predictor variables. In STATISTICA (StatSoft 2008) I used Mallows Cp selection criteria to select the simplest model that explained the most amount of variation. When a significant metal index effect was found ($P < 0.05$), best subset regressions were re-run using each of the individual metals and conductivity to identify which metal had the strongest effect on the community metric.

Finally, in ArcGIS, the interpolation tool 'kriging' was used to map the taxonomic richness and total densities of both *Deleatidium* and *Austroclima* across the Stockton Plateau and six streams on the coastal plain. Kriging is a geostatistical technique that is used to interpolate a value (in this case taxonomic richness or total densities of both *Deleatidum* sp. and *Austroclima* sp.) across a defined spatial extent by predicting the unmeasured value from measured values of points nearby (Jansen et al. 2002). The values derived from kriging are based on generalised least-squares regression algorithms. More detailed mathematical explanations are provided by Goovaerts (1997).

Results

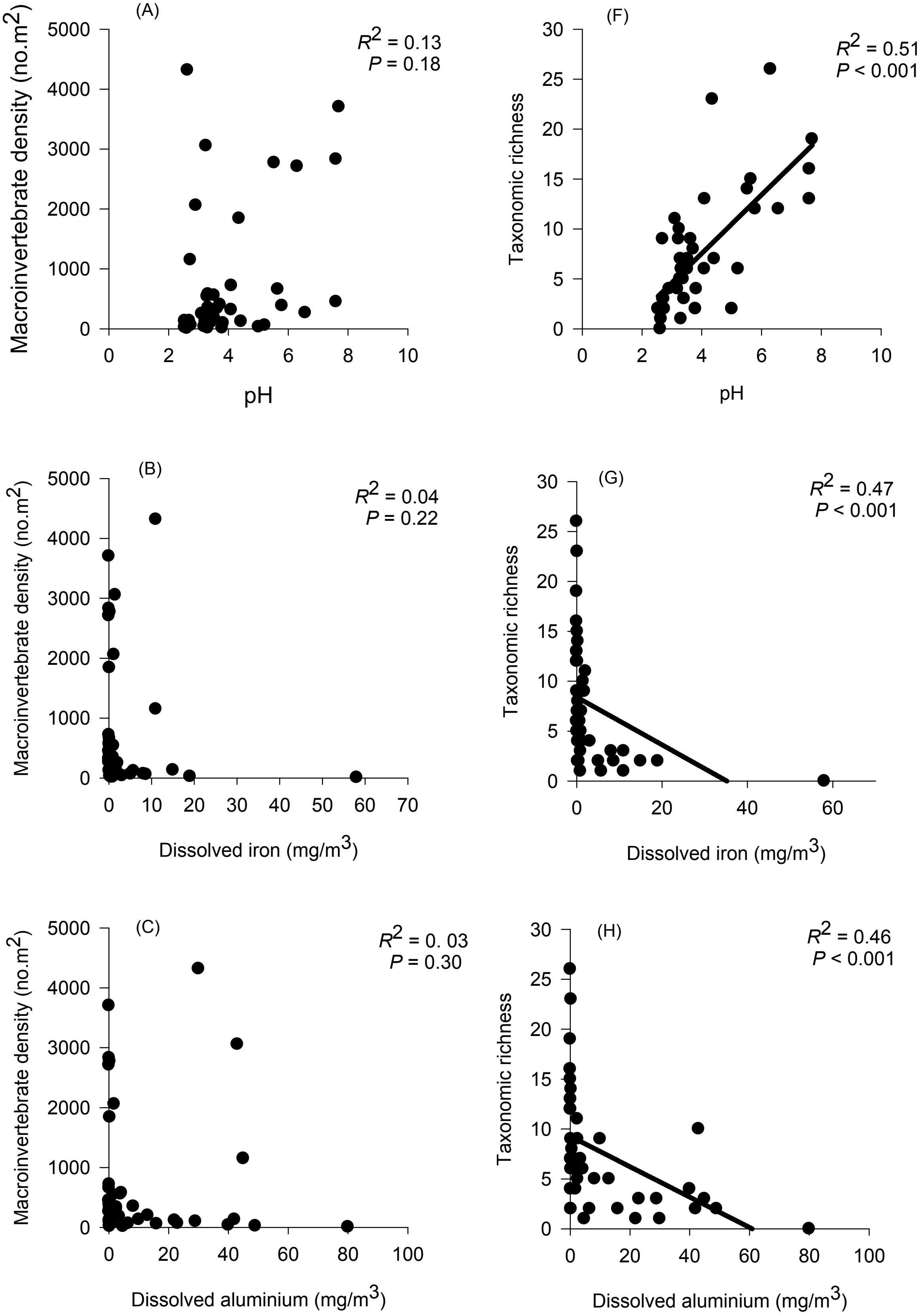
A total of 62 taxa were collected from 45 streams draining the Stockton Plateau and associated coastal catchments. These 45 streams were divided into three categories: reference, mildly and severely impacted (Table 2.1). Reference sites were naturally acidic (median pH 5.6, range 4.1-7.8) and had relatively diverse macroinvertebrate communities. These streams consisted of mayflies (*Deleatidium* and *Austroclima*), stoneflies (*Austroperla*, *Zelandobious* and *Zelandoperla*) and caddisflies (*Psilochorema*, *Pycnocentria* and *Rakiura*). In contrast, streams severely impacted by AMD had a median pH of 2.75 (range: 2.5-3.5) and high concentrations of metals (Table 2.1). Communities in severely impacted streams were dominated by Diptera (Chironomidae and Empididae) with the stonefly *Spaniocercoides philpotti* and caddisflies *Oxyethira albiceps* and *Psilochorema* sp. also being found at mildly impacted sites. Regression analysis between number of total taxa and pH and all metal values revealed a significant, negative relationship (Figure 2.2f-j). In contrast there was no significant effect of pH on macroinvertebrate density.

Local-scale drivers of macroinvertebrate community composition

RDA analysis of community composition revealed two significant axes which cumulatively explained 26 % of the variation (Axis 1 explaining 22 %, Figure 2.3). Forward selection identified two environmental parameters (pH and the metal index) that strongly influenced community composition. pH and the metal index were most strongly correlated with axis 1 ($R^2 = 0.83$, $P < 0.05$; $R^2 = 0.19$, $P < 0.05$, respectively). Reference streams occurred to the right side of axis 1 (high pH, low metals) whereas severely and mildly impacted streams clustered together on the left of axis 1 (high metal index and low pH) (Figure 2.3). There was a tendency for mildly impacted streams to occur between severely impacted and reference streams. Best sub-set regression identified the variables that best explained a range of macroinvertebrate community variables (Table 2.2). The metal index was the most useful predictor for 10 out of 17 community metrics. Of these 10 predictors, nine of them were significant at $P < 0.05$. Total taxonomic richness was the only community metric that was significantly predicted by both the metal index ($P = 0.043$) and pH ($P = 0.024$). QMCI and MCI scores were both affected by temperature, while pH was a significant biological predictor for QMCI ($P < 0.001$).

Table 2.1: Mean (±SE) (except pH which are median values) values for chemical data for streams draining the Stockton Plateau. Streams were sampled once between November 2007 and January 2008. Streams were categorised into three impact categories which were based on expert site knowledge.

Impact category	n	pH	Conductivity (25ºC μS cm^{-1})	Temperature (ºC)	Turbidity (NTU)	Suspended Solids (mg/l)	Diss. Fe gm^{-3}	Diss. Al gm^{-3}	Diss. Zn gm^{-3}	Diss. Ni gm^{-3}
Reference	14	5.6	56.56 (10.96)	15.28 (0.68)	0.55 (0.08)	2.00 (0.00)	0.10 (0.02)	0.17 (0.003)	0.01 (0.001)	0.0007 (0.00009)
Mildly	11	3.63	142.98 (25.66)	16.65 (1.09)	0.81 (0.20)	2.75 (0.38)	0.51 (0.18)	1.74 (0.44)	0.05 (0.01)	0.01 (003)
Severely	19	2.75	705.71 (103.55)	16.68 (0.66)	3.85 (1.39)	4.02 (0.68)	8.21 (3.03)	26.3 (4.66)	0.54 (0.14)	0.10 (0.03)


(A)
Macroinvertebrate density (no.m^2)
5000
4000
3000
2000
1000
0
0
2
4
6
8
10
pH
R^2 = 0.13
P = 0.18
(F)
Taxonomic richness
30
25
20
15
10
5
0
0
2
4
6
8
10
pH
R^2 = 0.51
P < 0.001
(B)
Macroinvertebrate density (no.m^2)
0
10
20
30
40
50
60
70
Dissolved iron (mg/m^3)
R^2 = 0.04
P = 0.22
(G)
Taxonomic richness
0
20
40
60
Dissolved iron (mg/m^3)
R^2 = 0.47
P < 0.001
(C)
Macroinvertebrate density (no.m^2)
0
20
40
60
80
100
Dissolved aluminium (mg/m^3)
R^2 = 0. 03
P = 0.30
(H)
Taxonomic richness
0
20
40
60
80
100
Dissolved aluminium (mg/m^3)
R^2 = 0.46
P < 0.001

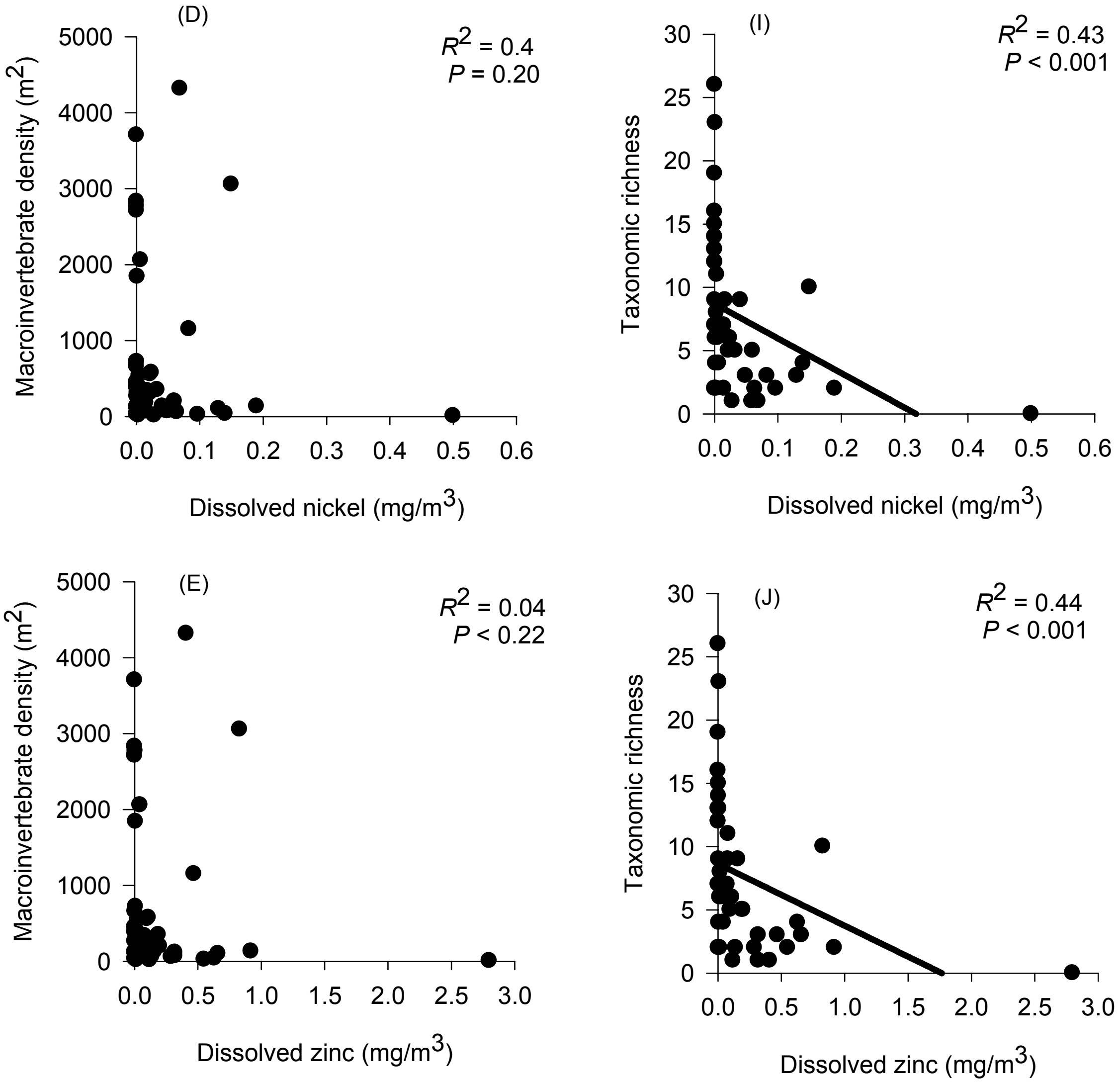


Figure 2.2: Scatter plots showing the relationships between pH and metals on macroinvertebrate density (a-e) and taxonomic richness (f-i) for 45 streams on the Stockton Plateau sampled once between November 2007 and January 2008. Relationships were significant at $P < 0.05$.

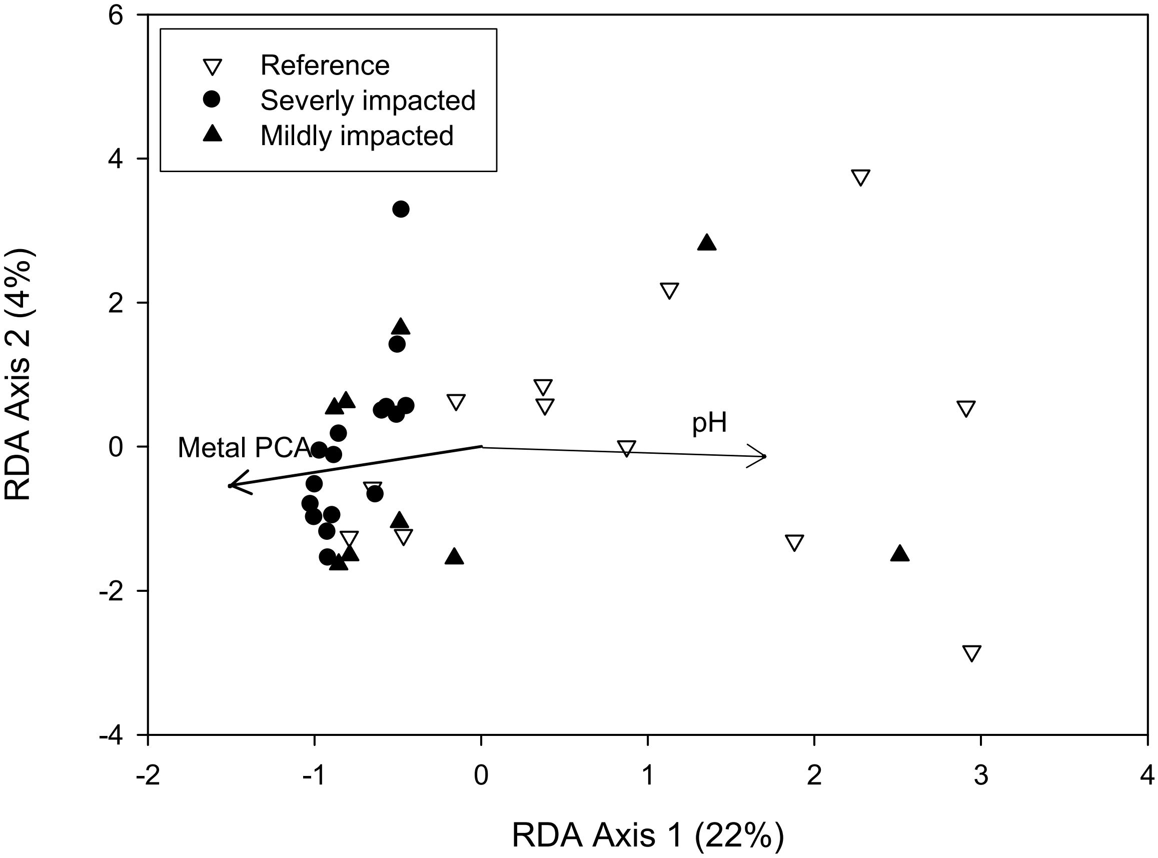


Figure 2.3: Redundancy analysis (RDA) for macroinvertebrate community composition in 45 streams on the Stockton Plateau and two significant ($P < 0.05$) environmental predictors that were selected by forward stepwise regression. Sites are divided into three impact categories. The length of the environmental gradients is proportional to correlation coefficients. The metal index is described in the text.

Table 2.2: Results of Best Subsets regression between the physical and chemical condition of water and macroinvertebrate community response variables for 43 streams on the Stockton Plateau sampled on one occasion between November 2007 and January 2008. (Two sites were removed from the data set because one was drive and one was devoid from any macroinvertebrate life).Values are significant at P <0.05. F values and slopes are also provided. Significance codes: n.s. non-significant; * $P < 0.05$; ** $P < 0.01$; *** $P < 0.001$; **** $P < 0.0001$.

Community metric		Best model		Temp. (°C)			pH			Metal index			Turbidity (NTU)		
	R^2	F	P	F	P	Slope	F	P	Slope	F	P	Slope	F	P	Slope
QMCI	0.67	25.46	****	2.37	0.13	-0.10	13.00	***	0.75						
MCI	0.49	18.93	****	4.98	0.03	-2.38				27.35	****	-16.70			
Margalef index	0.58	26.67	****				4.00	0.05	0.19	6.78	0.01	-0.37			
Berger-Parker index	0.36	10.75	***				2.33	0.14	-0.11	14.77	***	-0.42			
Total taxa	0.56	24.88	****				5.34	0.02	1.72	4.40	0.04	-2.25			
Total EPT Taxa	0.59	27.60	****				44.08	****	1.97						
% EPT	0.66	17.37	****	22.81	***	-0.06				16.7	***	-0.17	8.26	0.01	-0.09
% E	0.49	12.07	****	13.63	***	-0.04				6.68	0.01	-0.08	3.04	0.09	-0.08
% P	0.46	10.97	****	13.63	***	-0.04				6.68	0.01	-0.08	3.04	0.09	-0.04
% T	0.19	4.59	*	3.42	0.07	-0.03							7.05	0.01	-0.09
% Chironomids	0.57	16.92	****	30.6	0.09	2.32				22.75	***	20.12	5.54	0.02	7.84
% Mollusc	0.08	3.34	0.07				3.34	0.08							
% Shredders	0.15	7.33	*				8.79	0.01		7.33	0.01	-2.81			
% Filter feeders	0.18	8.9	**				3.34	0.08							
% Grazers	0.08	3.34	0.08										1.60	0.21	3.88
% Collector-browsers	0.04	1.60	0.21												
% Predators	0.18	4.15	0.02							4.12	0.49	3.65	7.03	0.01	2.22

Regional scale drivers of macroinvertebrate community composition

Kriging analysis in ArcGIS revealed distinct spatial distributions for taxonomic richness and the total densities of the two dominant mayfly taxa (*Deleatidium* and *Austroclima*). Taxonomic richness was most abundant around the periphery of the active mine area. These periphery areas include the coastal plain streams to the west and streams to the east (T69 and T35) (Figure 2.4). Smaller areas of moderate taxonomic richness were located in the north-eastern corner (e.g., Repo Stream and MT6) in addition to sites M1 and M2 Stream 2 and Plover 2. The lowest richness was found in areas with active or historic AMD, or areas with sediment issues from drill tracks or haul roads, or from naturally acidic catchments. Densities of mayflies demonstrated a similar gradient to taxonomic richness as densities were higher to the west on the coastal plain and the north-east (i.e., Repo Stream) (Figure 2.5). The densities of mayflies had a more polarised position on the periphery of the mine compared to taxonomic richness. Many of these sites that are closer to the active mining area (e.g. WW 1, 2, 3) had higher numbers of diptera and seemingly AMD tolerant caddisflies.

pRDA ordination was used to test the relative importance of environmental and spatial factors in influencing macroinvertebrate community composition and abundance. Environmental variables and geographic landscape position were both significant predictors ($P < 0.05$) of macroinvertebrate community composition (Figure 2.6). Environmental variables were the most important factors which explained the largest proportion of variation (24.5%). However, when the influence of environmental variables on macroinvertebrate communities was removed, geographic landscape position by itself accounted for 6.9 % of total community variation. The shared contribution of environment and landscape position accounted for 6.3% of the variation in community composition.

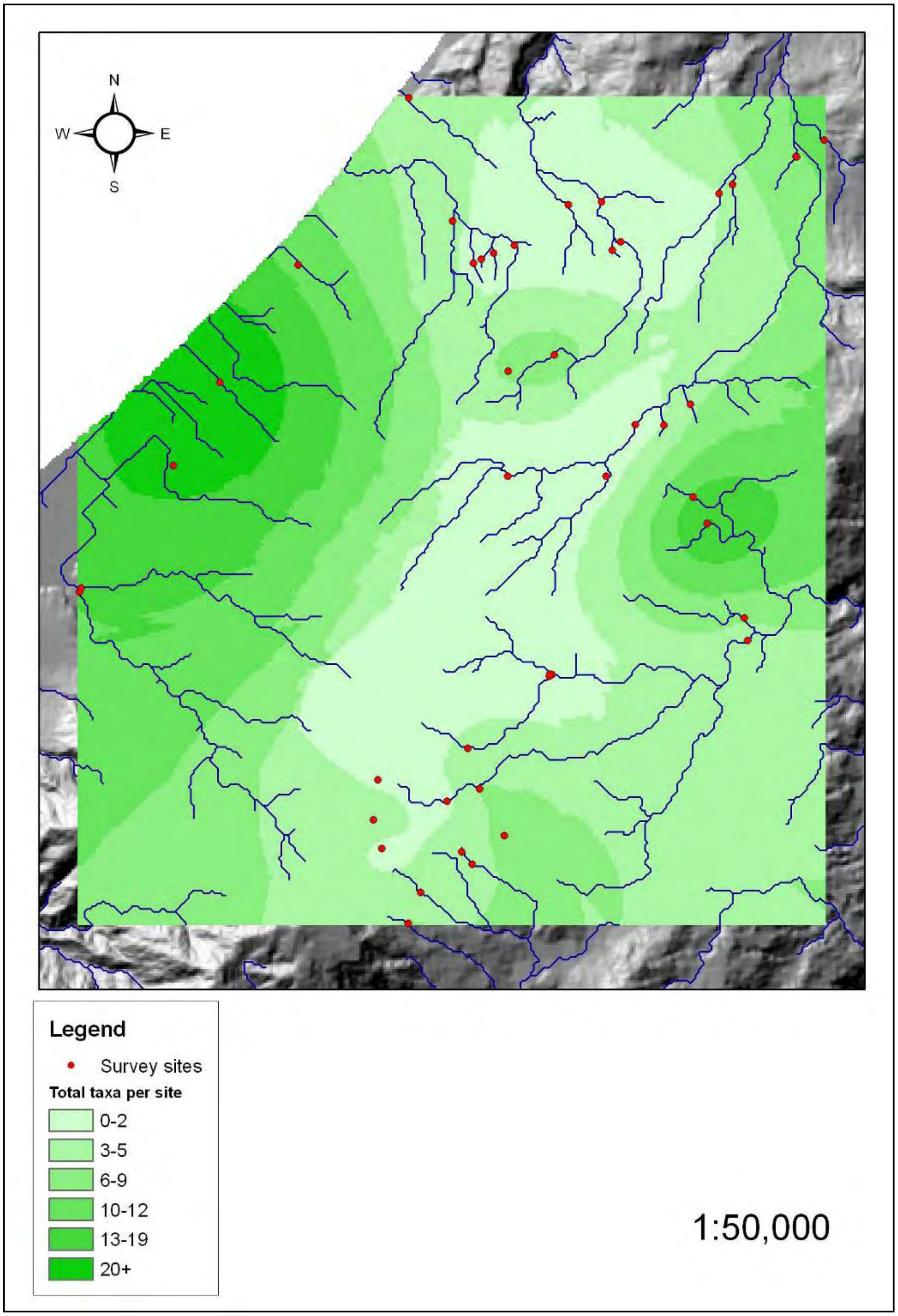


Figure 2.4: Kriged values of total taxa in 45 streams draining the Stockton Plateau calculated in ArcGIS. Survey sites are shown as purple dots while kriged values are shown in green. Darker values have higher taxonomic richness.

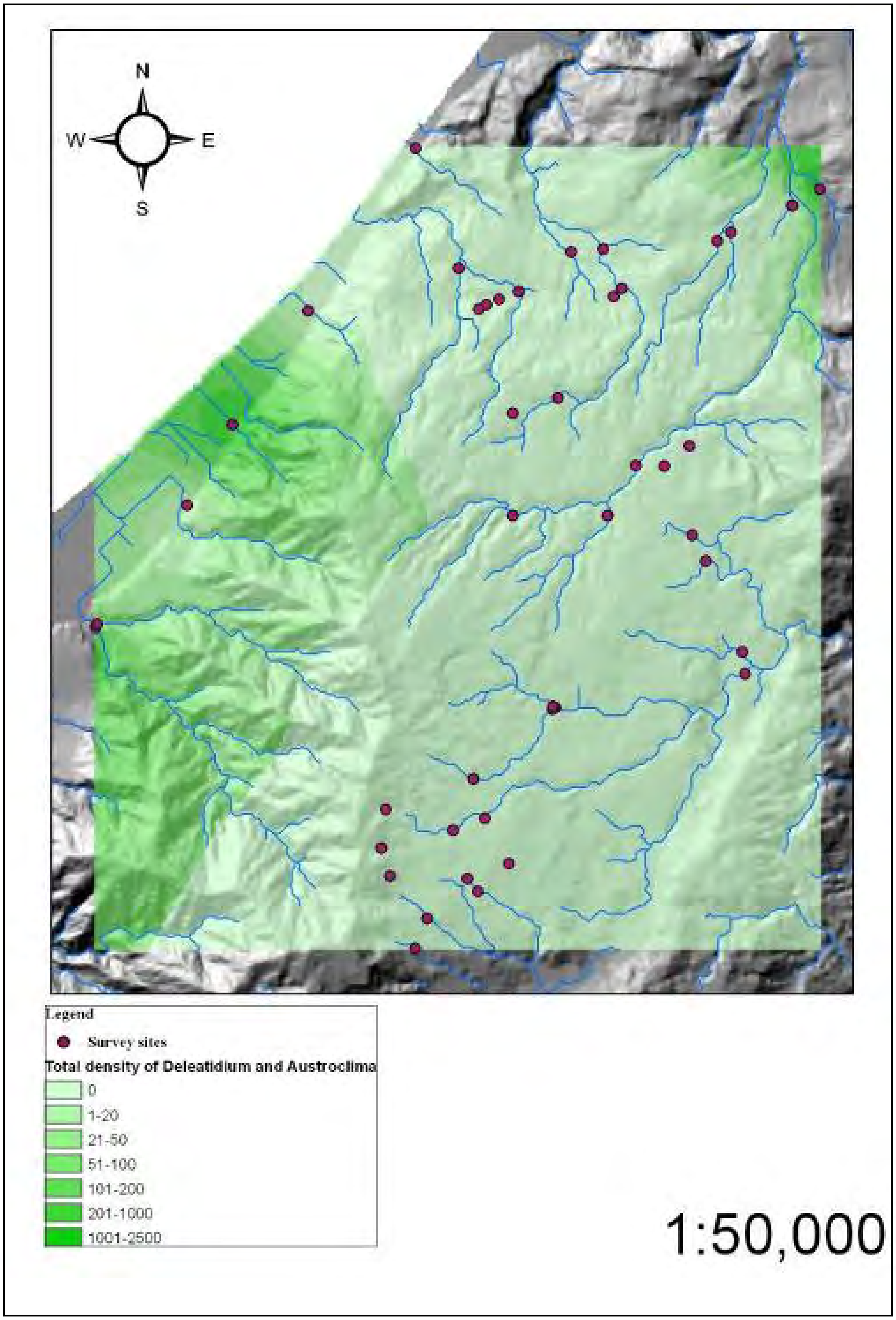


Figure 2.5: Kriged values of total density for both *Deleaditium* and *Austroclima* in 45 streams draining the Stockton Plateau calculated in ArcGIS. Survey sites are shown as purple dots while kriged values are shown in green. Darker values have higher densities.

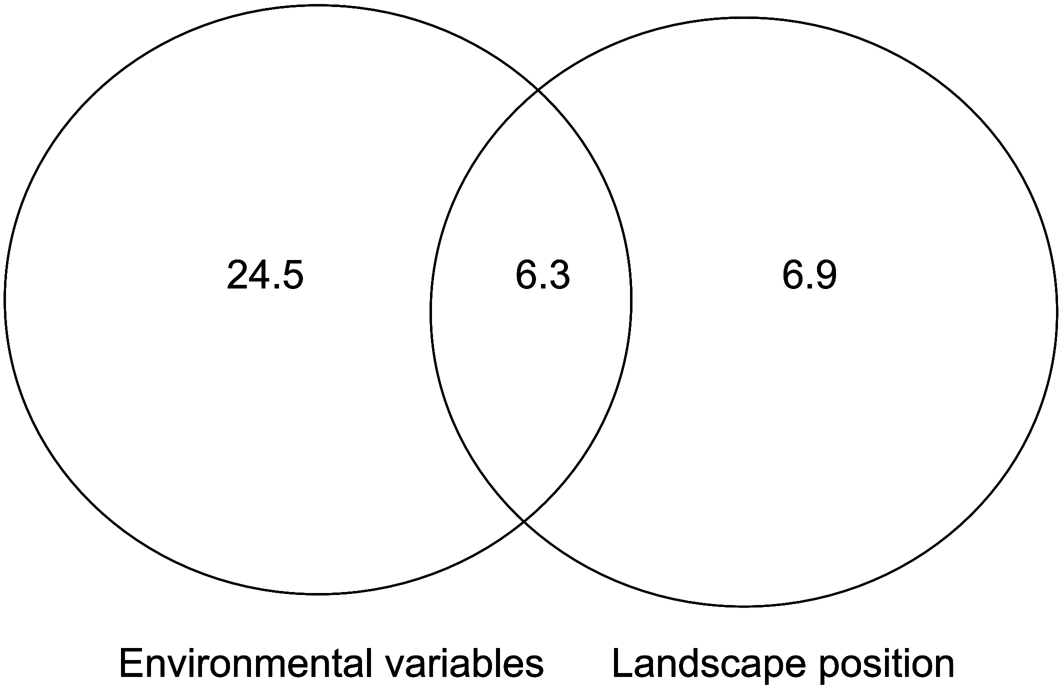


Figure 2.6: Venn diagram displaying the shared and independent explanation of environmental variables and landscape position that explains macroinvertebrate communities on the Stockton Plateau. The values were derived using a partial redundancy analysis (pRDA).

Discussion

Effect of environmental variables on macroinvertebrate community composition

This study found that environmental variables explained the greatest amount of variation in macroinvertebrate community structure on the Stockton Plateau. In particular, pH and the metal index were the primary variables responsible for influencing macroinvertebrate communities. Streams draining the Stockton Plateau generally have naturally low pH and low metal concentrations. However, the presence of open cast coal mining and the formation of AMD has lowered pH further and compounded this by increasing metal concentrations. The result is a complex environmental gradient caused by pH, elevated metal concentrations, sediment and iron precipitate. Along this environmental gradient, macroinvertebrate community composition changed with mayflies (*Deleatidium* and *Austroclima*) dominant at reference sites being replaced by chironomids at the most polluted sites. This finding is consistent with previous studies of AMD impacted streams on the West Coast of New Zealand’s South Island (Winterbourn and Collier 1987, Collier et al. 1990, Winterbourn and McDiffett 1996, Winterbourn 1998, Winterbourn et al. 2000a, Harbrow 2001, O'Halloran et al. 2008). Similarly, a switch in macroinvertebrate community composition from being dominated by sensitive EPT taxa to chironomids was also common among international acidification studies (e.g. Courtney and Clements 1998, Geurold et al. 2000, Berezina 2001, Cherry et al. 2001, Battaglia et al. 2005, Lin et al. 2005, Petrin et al. 2007a, Dsa et al. 2008, Petrin et al. 2008, Pond et al. 2008).

In contrast, the pH thresholds shown in overseas studies tend to be much higher than the pH values of my study. In naturally acidic streams, pH had a median value of 5.6 while AMD impacted streams had a median pH of 2.8. Naturally acidic streams are common on the West Coast of the South Island. This natural acidity is derived from humic and fulvic acids leaching into the stream from the catchment (Collier et al. 1990). Despite lower pH, these streams can still maintain diverse and complex macroinvertebrate assemblages (Collier et al. 1990, Winterbourn and McDiffett 1996, Winterbourn 1998, Harbrow 2001). Similar findings have been reported in Sweden where naturally acidic and anthropogenically acidic streams were compared (Dangles et al. 2004, Petrin et al. 2007a, Petrin et al. 2007b). In fact, of the 37 most widespread aquatic taxa in Westland streams, 92 % of them were found in streams with pH < 5.0 and 65 % of taxa were found at sites with pH < 4.5 (Collier et al. 1990). The ability of these taxa, including the mayfly *Deleatidium*, to survive these acidic conditions has been attributed to likely adaptation to low pH (Collier et al. 1990, O'Halloran et al. 2008). This hypothesis is also supported by overseas studies (Raddum and Skjelkvale 1995, 2001, Dangles et al. 2004, Petrin et al. 2007a, Petrin et al. 2008).

Given our finding of relatively diverse and abundance macroinvertebrate communities in naturally acidic streams on the Stockton Plateau, I suspect that AMD predominantly affects macroinvertebrates through its synergy with heavy metals. Metals have also been widely shown to have negative effects on macroinvertebrate communities (Maret et al. 2003, Soucek et al. 2003, Clark and Clements 2006, Doi et al. 2007, MacCausland and McTammany 2007, De Jonge et al. 2008, Van Damme et al. 2008, Iwasaki et al. 2009).

The anthropogenically related increase of metal concentrations in streams with naturally acidic pH may be particularly important given the apparent adaptation of macroinvertebrates to low pH (Collier et al. 1990, Raddum and Skjelkvale 1995, Winterbourn and McDiffett 1996, Raddum and Skjelkvale 2001, Dangles et al. 2004, Petrin et al. 2007a, Petrin et al. 2008). For example, Winterbourn and McDiffett (1996) found that the ability of Westland macroinvertebrates to tolerate low pH was dependent on low metal concentrations. When these metal concentrations increase as a result of AMD discharges, macroinvertebrate communities can be markedly altered. In streams impacted by hard rock mining in the American northwest, macroinvertebrate communities downstream of mines had significantly lower densities and taxonomic richness when compared to upstream, un-impacted reaches (Maret et al. 2003). This was attributed to significant increases in metals concentrations. pH was not an issue at these sites as they remained circum-neutral along the entire network (Maret et al. 2003). Similarly, Courtney and Clements (2002) surveyed five metal-polluted sites and one reference reach, which all had similar pH levels (6.6-7.4). They found that metal polluted water had significantly reduced macroinvertebrate abundance, particularly mayflies and stoneflies. Despite the demonstrated deleterious effects of metal pollution on macroinvertebrates, the effects can vary with the size and development stage of individual species. An assessment of macroinvertebrate density and size distribution in the Arkansas River, Colorado, by Clark and Clements (2006) was conducted in early spring and late summer. This was timed to coincide with early and late stage larval development. They found that significant mortality occurred for the mayfly *Rhithrogena hageni* in spring when larvae were in their early development stage. In contrast, no significant mortality was observed in summer when they were in their late development stage. The ability of larger individuals to survive in these metal polluted systems may account for the population's survival (Clark and Clements, 2006).

The influence of environmental and spatial factors

Despite environmental variables being the dominant factor controlling community structure, the position of a stream relative to other streams within the landscape can also exert a significant influence. This spatial effect may be best understood in terms of regional filters. Filters are essentially a series of conditions that an aquatic adult insect must pass before it can colonise a stream. Such conditions include the dispersal abilities of insects, the regional climatic conditions, vegetation, inter-stream isolation and topography (Collier and Smith 1997, Palmer et al. 1997, Briers et al. 2003, Wilcock et al. 2003, Caquet et al. 2007, Vanschoenwinkel et al. 2008). At the local scale, filters can include water chemistry, habitat availability and food availability at a particular stream (Palmer et al. 1997). On the Stockton Plateau, filtering at the regional scale is most likely to be driven by macroinvertebrate dispersal abilities and dispersal constraints. Each of these constraints may influence meta-community structure and dynamics on the Stockton Plateau for a variety of reasons. For example, due to the influence of AMD, some streams may become ecologically isolated because a stream that once acted as a stepping stone for dispersal to a distant site is now too polluted to function in that role (Lowe 2002, Caquet et al. 2007, Masters et al. 2007).

On the Stockton Plateau, the supply of recolonists through inter-stream dispersal may be limited by the type of vegetation on the plateau (such as forest, scrub, or open bedrock pavement) that may potentially limit lateral dispersal (Collier and Smith 1995, 1997, Palmer et al. 1997, Petersen et al. 1999). For example, Collier and Smith (1995) found that streams with different riparian forest cover had different aquatic adult species composition while willow forest had higher aquatic insect abundances dispersing into it than pine forest and native forest. Furthermore, Petersen et al. (1999) found significantly more *Leuctra fusca* and *Nemurella pictetii* at greater lateral distances from woodland streams than in open grassland. In contrast Delettre and Morvan (2000) found that chironomid dispersal was restricted in hedgerows, but free in open farmland. Uniquely characteristic of the Stockton Plateau is the fact that many of the streams are in heavily incised gullies forming mini-canyons (Plate 2.1). These incised gullies may function as dispersal conduits forcing adult insects to disperse longitudinally as opposed to laterally. The potential effect of this is that macroinvertebrates are discrete populations in space with limited connection to other streams. For example, connectivity between sites through dispersal could explain why the relatively close streams such as Repo and MT6, and T69 and T35, had very different macroinvertebrate communities despite similar water chemistries. All these streams had distinct macroinvertebrate communities, which I suggest was the result of complex terrain restricting dispersal; all sites were in incised gullies restricting migration to the stream corridor itself. In contrast, streams WW1, WW2, and WW3 had similar water chemistry, similar macroinvertebrate composition and were in open, less complex terrain.

Plate 2.1: A highly incised stream (MT6) that is typical of almost all streams draining the Stockton Plateau.

Finally, given the predominance of westerly winds in the summer, there could be a preferential dispersal gradient from west to east on the plateau. The effects of wind are often cited as an influence of insect directional flight and effectiveness (e.g. Waringer 1991, Caceres and Soluk 2002, Briers et al. 2003) All of these factors probably influence the distribution of macroinvertebrates across the landscape. Unfortunately, I was not able to quantify these filtering effects directly, so I am only able to speculate on their importance. Rather, I will focus on the importance of landscape position.

Understanding meta-community theory is important to assist in the recovery of macroinvertebrate communities after AMD has been treated as it allows practitioners to identify some potential barriers or influences that may control macroinvertebrate recovery. For example, in a remediated Welsh stream, Masters et al. (2007) found that macroinvertebrate communities in streams treated for AMD recovered quickly as a result of a series of natural streams acting as stepping stones to the remediated reach. Consequently, failure to appreciate the importance of meta-community dynamics may negatively affect remediation success. Four meta-community theories might be applicable to the distribution of species on the Stockton Plateau. Patch dynamics assumes that patches are identical and can contain a macroinvertebrate community that is limited by dispersal. Conversely, mass effects assume that immigration and emigration influence local community structure and that species that are poor competitors are salvaged from local extinction by dispersal from systems where the species is abundant. Species sorting conjoins both of the above paradigms and advocates that resource gradients or patch types create strong differences in the local composition of communities. The resultant community structure is thus a function of both dispersal abilities and patch/resource quality. The neutral perspective takes a more random view as all species are similar in their competitive abilities, dispersal abilities and fitness and the community composition is derived from the probabilities of species loss and gain (Leibold et al. 2004).

In the context of this study, the pRDA (Figure 2.6) found that environmental variables explained more variation in macroinvertebrate community composition than a community's geographic position within the landscape. Thus, using the definitions provided by Leibold et al. (2004), it would appear that given the combination of complex topography (incised gullies) and the influence of local variables on species distribution on the Stockton Plateau that the most applicable paradigm is likely to be species sorting. Species sorting assumes that community composition is determined by local abiotic conditions, which in this landscape is an AMD impact gradient. This is compounded by the dispersal abilities and constraints of individual taxa which explain the significant, but less dominant influence of dispersal. A similar conclusion was reached by Sanderson et al. (2005), who noted that pH and habitat structure were the primary influences on species composition. The secondary factor influencing community composition in their study was the species composition of surrounding streams and their dispersal abilities. Similarly, my data suggest that two layers of filters determine community structure in Stockton Plateau systems, similar to those proposed by Palmer et al. (1997). This conclusion is also supported by a meta-analysis of 158 data sets that revealed the majority of meta-communities were structured by a combination of both species-sorting and mass effects (Cottenie 2005). Similar results were reported for pond systems where environmental variables were the primary drivers of community structure though high dispersal rates of some taxa from surrounding patches also influenced their presence (Urban 2004). This suggests that species sorting and mass effect paradigms complement each other despite the hierarchical nature between local scales (environmental gradients) and regional scales (geographic landscape position). However, after AMD has been treated, it is possible that the dominant force allowing for the recovery of the macroinvertebrate communities will become species sorting. But this paradigm will be constructed differently in the remediating communities than the degraded communities because in theory all streams should have similar water chemistry. Therefore, the effects of pH and metal toxicity should be eliminated and the first level of filters such as dispersal constraints will influence the success of re-colonisation by AMD sensitive taxa.

Implications of environmental variables and landscape position for the remediation of AMD impacted streams and potential sources of colonists.

This study has demonstrated that in an AMD-impacted landscape, environmental variables (water chemistry) were the prevailing influence on macroinvertebrate community structure, while stream position within the landscape is a secondary structuring force. However, provided that there is successful treatment of AMD, it is highly probable that this condition will change, and environmental variables may become less important. This is because all streams treated for AMD should end up with broadly similar water chemistry to reference sites. Consequently, it is likely that stream position within the landscape will exert a greater influence on structuring macroinvertebrate diversity and community composition.

The finding that the spatial arrangement of streams within a landscape has an important influence on macroinvertebrate community composition is of vital importance to stream remediation, and demonstrates the importance of not relying on the “field of dreams” (i.e. if you build it, they will come) hypothesis. Specifically, because landscape position can also influence macroinvertebrate community composition, it cannot be assumed that colonists will automatically return after the stressor has been removed. It is essential to the success of any remediation project that dispersal is considered. Such factors include the proximity of source populations and dispersal constraints as proposed by Palmer et al. (1997). Consequently, a more suitable approach may be to remediate AMD impacted streams that are in close proximity to source populations, which can include intact tributaries, headwaters, or other nearby stream reaches. Once these remediated reaches begin to resemble natural communities, these sites can be used as stepping stones for colonists dispersing to other remediated reaches as shown by Masters et al. (1997). This is discussed in more detail in my final chapter.

Chapter Three

Sources of aquatic insect colonists to remediated streams: the relative importance of macroinvertebrate drift and adult insect flight

Introduction

Streams that have been disturbed by natural events or remediated from human induced degradation can recover quickly provided that there are sources of colonists (e.g. Matthaei et al. 1996, Matthaei et al. 1997, Wantanabe et al. 2000, Fowler 2002). There are four primary modes for which macroinvertebrates might use to recolonise systems. These are aerial flight, drift, movement through the groundwater and dispersal in the stomachs of other organisms (Williams and Hynes 1976, MacKay 1992, Winterbourn 2004). Dispersal by winged aquatic insects is an important part of their life-histories and allows adults to access streams and reaches that they might not normally be able to colonise (Bilton et al. 2001). It is also important because river systems are naturally fragmented and separated by terrestrial environments (Malmqvist 2002). Because of the unidirectional flow of streams, there is net loss of macroinvertebrates downstream which is compensated by aerial flight up-stream by adult insects. This is more commonly known as the colonisation cycle proposed by Müller (1954, 1982). Aerial flight is important because it allows adults to disperse between disconnected systems aiding gene flow and providing species opportunities to recolonise systems post disturbance (Palmer et al. 1996, Briers et al. 2003). However, due to potential limits of vegetation cover, climate and topography on dispersal between catchments (Collier and Smith 1995, 1998, Petersen et al. 1999, Briers et al. 2003, Petersen et al. 2004, Zickovich and Bohonak 2007, Zbinden et al. 2008) recovery of macroinvertebrate communities by aerial recolonisation can be slow. In contrast, Townsend and Hildrew (1976) found downstream drift accounted for 82 % of macroinvertebrates colonising disturbed substrate. In the south branch of the Waimakariri River in the South Island, Kramer (1982) investigated the pathways of colonists to disturbed substrate. He concluded that drift accounted for 43 % of colonists while adult flight only accounted for 13 %. Since then a number of studies have concluded that drift is the primary mode of recolonising naturally disturbed reaches (Matthaei et al. 1996, Matthaei et al. 1997, Fenoglio et al. 2002, Duran 2006, Zbinden et al. 2008).

The primary modes of re-colonisation of streams that have been remediated from human induced degradation are likely to be from either aerial inter-catchment dispersal or downstream drift. Initial studies of aerial dispersal investigated upstream and downstream flight patterns to test of Müller's (1954, 1982) colonisation cycle (e.g. Bird and Hynes 1981, Peckarsky et al. 2000, Winterbourn and Crowe 2001). However, flight along a stream continuum may be of limited benefit to remediated streams. Rather lateral, inter-catchment dispersal may be more important. A number of studies have investigated lateral dispersal using direct trapping methods (such as malaise trapping or sticky trapping) have found an exponential decline of adult aquatic insects with increasing distance away from the stream (e.g. Jackson and Resh 1989, Sode and Wiberg-Larsenm 1993, Collier and Smith 1995, 1998, Petersen et al. 1999, Petersen et al. 2004, Winterbourn 2005). Advances in genetic and isotope analysis technology, have allowed researchers to use indirect methods, such as isotope enrichment and genetic analysis, to investigate inter-catchment dispersal with more confidence (MacNeale et al. 2004). Such studies have shown that a small number of individuals are able to disperse large distances and provide sufficient recruits to populate a stream (Hughes et al. 1998, Hughes et al. 1999, Schultheis and Hughes 2005, Wilcock et al. 2007, Zickovich and Bohonak 2007, Zbinden et al. 2008). For example, in a mark-recapture experiment, McNeale et al. (2005a) labelled macroinvertebrates using a ^{15}N tracer and found that adult female stoneflies readily dispersed to an unconnected stream 500 m away. Using a similar stable isotope enrichment technique, Briers et al. (2005) also concluded that the stonefly *Leuctra inermis* dispersed distances ranging between 800 m and 1,100 m from their natal stream.

Opencast mining can have significant adverse effects on macroinvertebrate communities due to physical and chemical changes in the stream as well as the physical destruction of the terrestrial environment. To rectify this effect, mining companies are required to remediate aquatic and terrestrial ecosystems. However, an important component affecting the success remediation is the understanding of likely colonists sources (which I identified in Chapter 2 and will be further discussed in Chapter 5) as well as identifying the dispersal mechanisms that will enable recolonists to arrive. Consequently, I wanted to investigate lateral dispersal of adult aquatic insects and the effect that different vegetation had on lateral dispersal. Three vegetation types are common on the Stockton Plateau (riparian forest, scrub and open bedrock) and it seems likely forest might impede lateral dispersal while bedrock and scrub might enable adults to fly greater distances. Furthermore, open bedrock and scrub vegetation are likely to be the dominant the landscape in a disturbed mine site. In this study I also want to determine the relative importance of drift and downstream flight in supplying colonists to remediated streams.

Methods

Study Sites

This study was conducted at six streams on the Stockton Plateau, a gently undulating plateau 500-700 m a.s.l. and 21 km east-northeast of Westport, New Zealand (Figure 3.1). The Stockton Mine is New Zealand's largest opencast coal mine (Henley 2007), and is located on the Stockton Plateau. The plateau is characterised by a climatically extreme environment. The predominant wind direction is from the west and north-west and wind speeds can be high because the plateau rises steeply from sea level to the Mt Augustus and Mt Fredrick ridgeline. This ridgeline forces wind to ascend the 1,100 m steep western face of the plateau cooling and condensing the air producing orographic rainfall which exceeds 6,000 mm annually (Alarcon 1997). Due to the extreme climate and poor soil, vegetation is slow growing. As a result, much of the plateau is either exposed bedrock or low scrub with native riparian forest in sheltered ravines. On the exposed rock pavement, vegetation is sparse and areas of exposed bedrock are common. Where vegetation does grow it is stunted and dominated by prostrate manuka (*Leptospermum scoparium*), as well as grasses and mosses which generally only get to 30 cm in height. Manuka (which can grow up to 1.5 m) dominates the gently sloped areas of the plateau along with carpet grass (*Chinochloa australis*), wire rush (*Juncus* sp.) and tussock (*Chionochloa* sp). Lowland forest, dominated by southern rata (*Metrosideros umbellata*), mountain beech (*Nothofagus solarnadri v. cliffortioides*), mountain totora (*Phyllocladus alpinus*), yellow-silver pine (*Lepidothaminus intermedius*), pygmy pine (*Lepidothamnus intermedius*), *Dracophyllum densum* and *Dracophyllum politum*, is common in gullies and on the eastern facing slopes where it can grow to 4 m. This better growth is due to better drained soils and less exposure to the dominant westerly winds (Alarcon 1997, Theinhardt 2003).

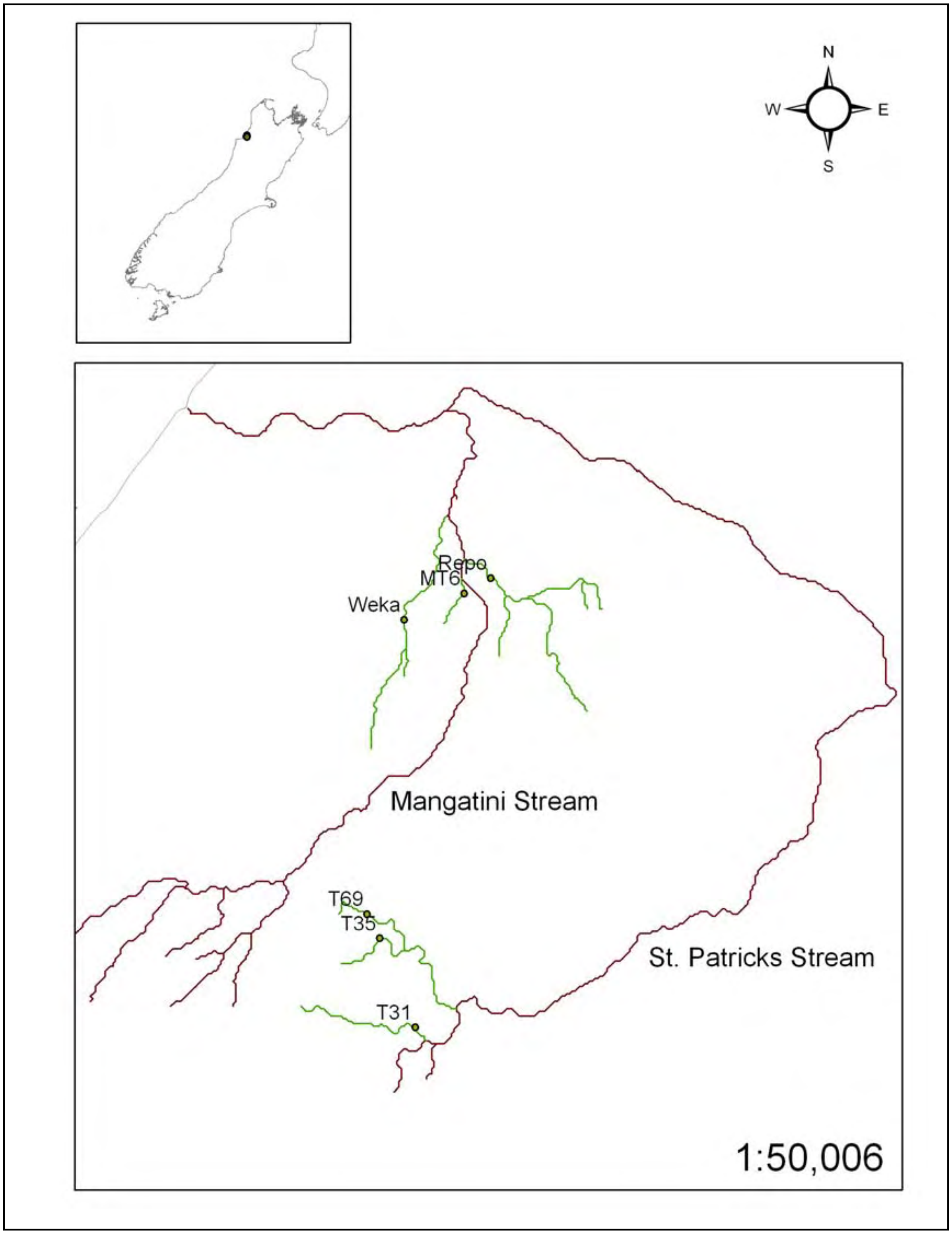


Figure 3.1: The location of the Stockton Plateau and the six study streams used for drift-net and aerial flight sampling.

Adult aquatic insect dispersal

Sticky traps were deployed to sample the relative importance of lateral and longitudinal dispersal of adult aquatic insects in three different vegetation types (open bedrock, scrub and native riparian forest) along a 500 m reach of one stream, T35, on the Stockton Plateau (Figure 3.1). T35 was the only stream selected for this study because it was unimpacted by AMD, had relatively high taxonomic richness (Appendix 1) and was the only stream where examples of the three vegetation types characteristic of the Stockton Plateau were present along the entire 500 m study reach. Sampling was conducted between November 2008 and January 2009, during which traps were deployed over four seven day periods.

To quantify lateral dispersal, I attached five replicate sticky traps within each of the three vegetation categories at five intervals (0, 5, 10, 20 and 30 m from the stream edge) perpendicular to one side of the stream (Figure 3.2a). At each distance from the stream, the five replicate traps were offset at least 1 m from each other, and traps for each lateral distance were offset from replicate traps on the subsequent lateral distance to prevent interference (Figure 3.2b). Each trap was constructed from a piece of clear acetate sheet (297 x 210 mm) stapled to plastic trellis (Plate 3.1) and suspended 0.8 m off the ground and coated in Tanglefoot™. Longitudinal trapping was also conducted within each of the three vegetation categories, between 5 and 10 m downstream of the lateral trapping (Figure 3.2a). Three sticky traps were set up at each site between 0.05 m and 2 m above the stream surface to capture aquatic insects flying longitudinally (Plate 3.2). The traps had a clear acetate sheet stable to each side of plastic trellis and were coated with Tanglefoot™ on both sides.

Every seven days, lateral and longitudinal sticky traps were collected and frozen, and fresh traps were deployed, giving a total of four trapping periods between November 2008 and January 2009. However, due to two separate flooding events, two of the four longitudinal sampling occasions were lost. In the laboratory, all Ephemeroptera, Plecoptera and Trichoptera (EPT) adults were identified to family level under a 10 x microscope using an unpublished key provided by Michael Winterbourn. The number of EPT adults captured over the multiple sampling occasions was pooled to give a total EPT catch for each trap between November 2008 and January 2009. These were then expressed as the total number of adult EPT captured per m^2.

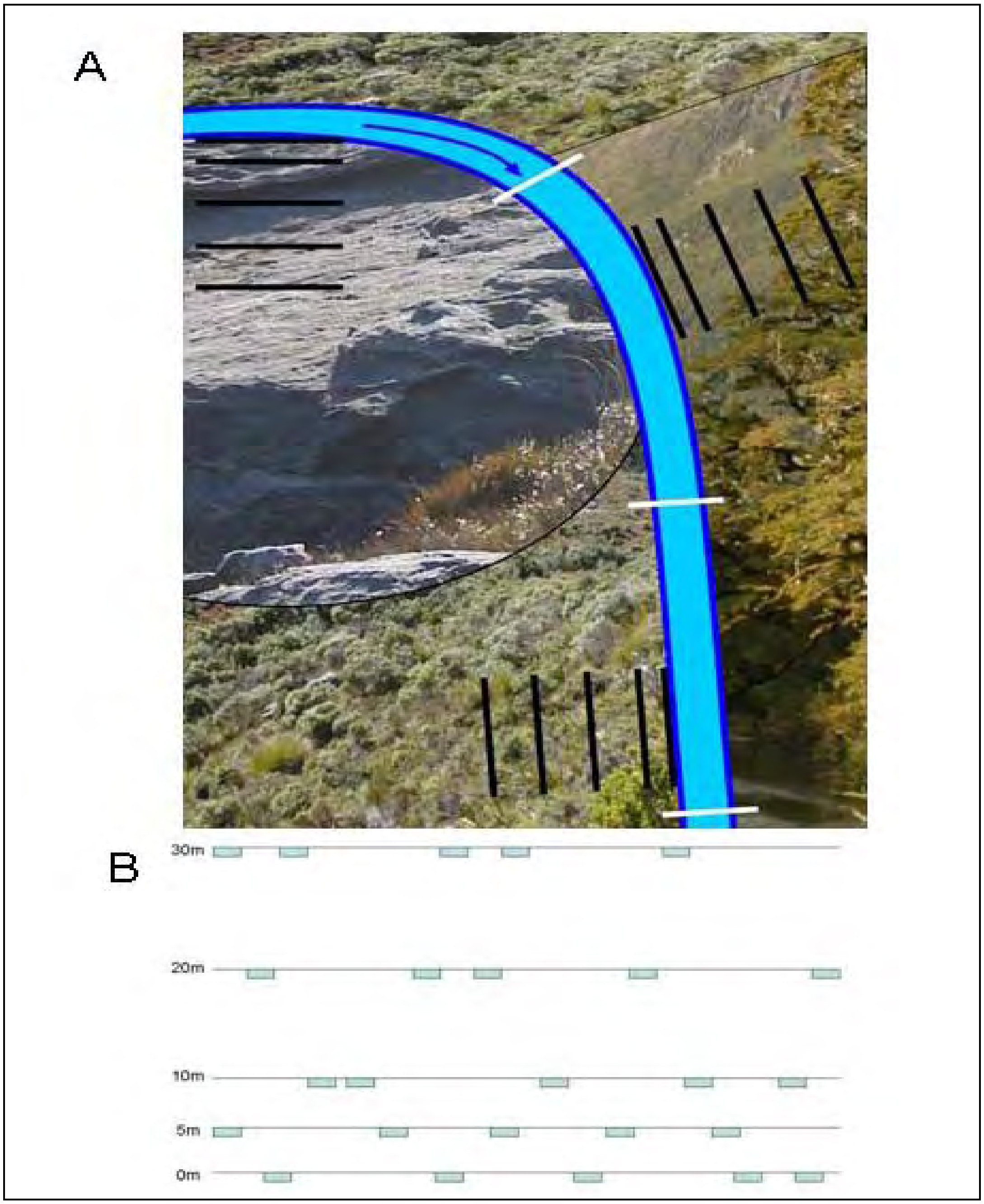


Figure 3.2: Schematic (a) shows how trapping lines (black lines) were deployed in three different vegetation covers at a single stream (T35) relative to the stream with the lateral traps shown in white lines across the stream. Schematic (b) shows how replicate sticky traps (blue boxes) were deployed long transects at each site.

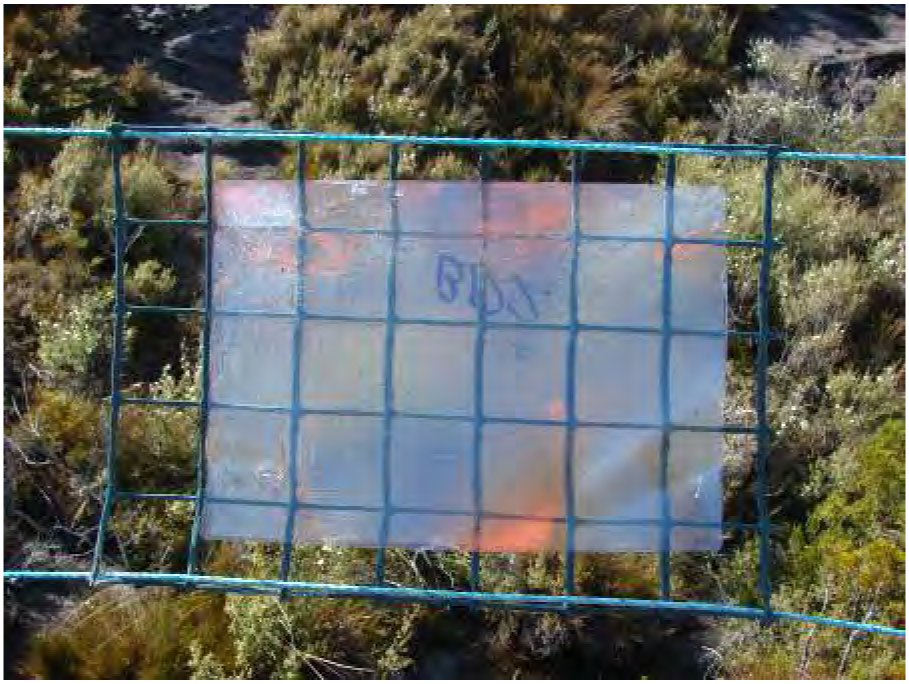


Plate 3.1. Sticky traps were constructed from clear acetate sheets stapled to plastic trellis.

Plate 3.2: Longitudinal traps established at three heights above the water surface at the scrub site.

To determine the predominant longitudinal flight direction of adult EPT taxa and the influence of vegetation, a two-way analysis of variance (ANOVA) was used to test the difference between upstream and downstream flight as well as the effect that vegetation type had. The density of EPT taxa per trap was calculated with the same method described above. The response variable (aquatic adult insect density) was checked for normality and if required, log transformed. All analyses were conducted in Statistica (Version 8) (StatSoft 2008) and significant values were detected at $P < 0.05$.

Analysis of covariance (ANCOVA) was used to determine if the number of adult EPT taxa captured changed with increasing lateral distance (co-variate) from the stream channel or from the influence of the three vegetation types (fixed factor). To calculate the numbers of adult EPT taxa for each replicate trap, adult EPT taxa were converted into densities by pooling all the data from the four sampling periods together and extrapolating this to density per square metre.

Benthic macroinvertebrate drift and downstream aquatic adult insect flight

To compare the relative importance of instream drift versus downstream aerial flight of aquatic adult insects for supply of colonists, benthic density, drift density and adult insect flight downstream were quantified in six streams, over three days in January 2009. The six streams; Repo Stream, MT6, Weka Creek, T31, T35 and T69 (Figure 3.1), were selected because they had high proportions of EPT taxa. The first three streams (Repo Stream, MT6 and Weka Creek) were all tributaries of the Mangatini Stream, and the latter three streams (T31, T35 and T69) are tributaries of St Patricks Stream (Figure 3.1). Both Mangatini and St Patrick's streams are currently under active remediation from AMD. In each of the six study streams, three replicating benthic Surber samples (0.04 m^2; 500-µm mesh) were collected from randomly selected locations in a 30 m reach that included at least one riffle and one run. Samples were preserved in 70 % ethanol in the field and returned to the laboratory. Three sticky traps were constructed by stapling a clear acetate sheet (297 x 210 mm) onto plastic trellis and coated with Tanglefoot™ on the upstream facing side and placed side by side (Plate 3.3). The sticky traps were suspended from the riparian vegetation about 2 cm above the water surface, upstream of where the Surber samples were taken and collected after 72 hours. Two drift nets (300 x

250 mm, 500-µm mesh) (Plate 3.4) were placed side-by-side in each stream where the flow was constricted, upstream of the sticky traps. The drift nets were cleaned of all macroinvertebrates every 12 hours and replaced over a 72 hour period. Drift samples were preserved in the field with 70 % ethanol.

Figure 3.3: Three sticky traps downstream of drift nets at Repo Stream on the Stockton Plateau.

Plate 3.4: Drift nets located at T31.

In the laboratory, the Surber samples and drift samples were washed through a 500-µm sieve before EPT taxa were counted and identified under a 10 x microscope to the lowest taxonomic level using the keys provided by Smith (2001) and Winterbourn et al. (2000b). Sticky traps were deforested and aquatic EPT adult aquatic insects were counted and identified under a 10 x microscope to family level using an unpublished guide provided by Michael Winterbourn.

Benthic macroinvertebrate density was calculated by extrapolating the mean density of the three Surber samples. Macroinvertebrate drift was expressed per 24-hour period. In each stream, the total number of aquatic macroinvertebrates from the EPT order from both drift nets were added together and then extrapolated to represent the entire width of the wetted channel. This was then converted to a density per square metre so it could be compared to aerial flight densities. Aquatic adult flight densities were calculated by taking the mean of the replicated sticky traps, extrapolating them out to a square metre and then dividing by three to convert to mean density per 24 hours. This was so the drift samples and sticky trap data had the same units of measure to determine whether drift or flight is more important. The difference between the number of individuals drifting or flying past a specific point in a 24 hour period was then tested using a one-way ANOVA in Statistica (Version 8)(StatSoft 2008).

Results

Insect fauna, flight direction and the influence of vegetation cover on longitudinal flight

A total of 100 individuals comprising adults from the EPT orders were caught over the four trapping periods, 75 % were caught on lateral traps, while the remainder were caught on longitudinal traps. Two-way ANOVA indicated that there were no significant differences in the density of adult EPT dispersing up- or downstream ($F_{1,17} = 0.856$, $P = 0.373$)(Figure 3.3), nor was there an effect of the type of riparian vegetation ($F_{2,17} = 0.891$, $P = 0.436$).

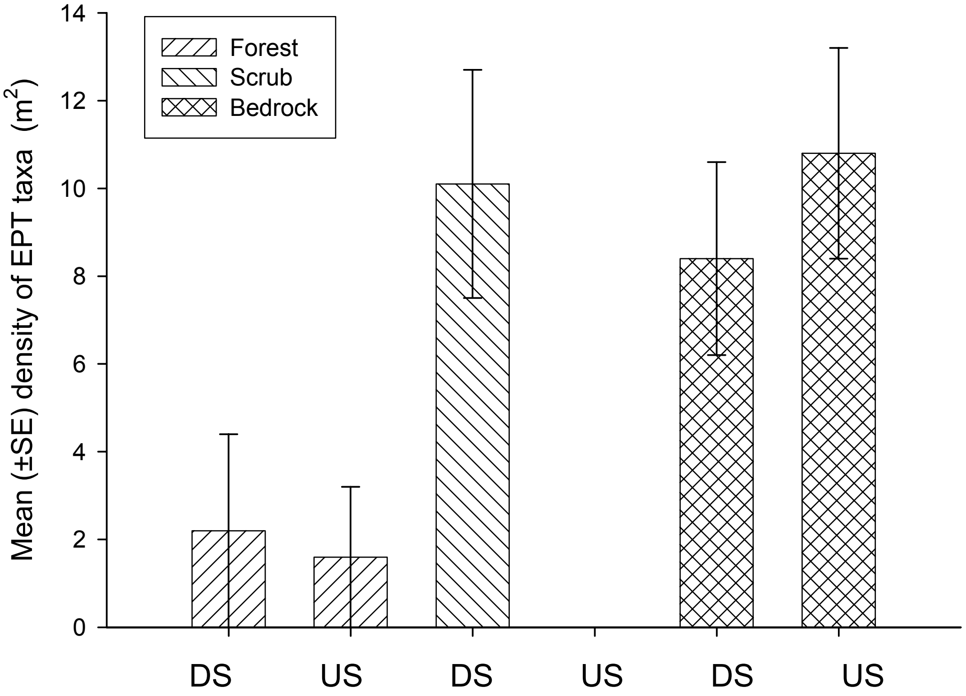


Figure 3.3: Mean (± 1SE) number of adult EPT taxa caught on upstream and downstream longitudinal sticky traps at T35 on the Stockton Plateau between November 2008 and January 2009. US = upstream; DS = downstream.

ANCOVA revealed that vegetation cover did not have any significant influence on the density of individuals dispersing laterally from the stream ($F_{2,\ 59}$ = 1.17, P = 0.318) (Figure 3.4). In addition, the density of EPT adults dispersing away from the stream edge did not decline significantly with increasing distance ($F_{1,\ 59}$ = 0.01, P < 0.99). Despite the lack of significant effects of vegetation and lateral dispersal distances, patterns were still evident. The density of adult insects in forest vegetation (Figure 3.4a) was very low compared to bedrock (Figure 3.4b). In contrast, scrub vegetation had relatively high densities of adult insects and mean densities decreased slightly with increasing distance (Figure 3.4c) which was in contrast to bedrock where mean adult insect density declined rapidly. Vegetation type may have also influenced the taxa dispersing. For example, in forest only Trichoptera (Hydropsychidae) were collected, in contrast to scrub and open bedrock which both had high proportions of Trichoptera (Hydroptilidae, Oeconesidae and Hydrobiosidae) and small proportions of Plecoptera (Gripopterygidae). Ephemeroptera (Leptophlebiidae) were only caught on sticky traps in scrub vegetation (Figure 3.5). 94 % of all Gripopterygidae were caught in the first 10 m from the stream edge in open bedrock and scrub vegetation.

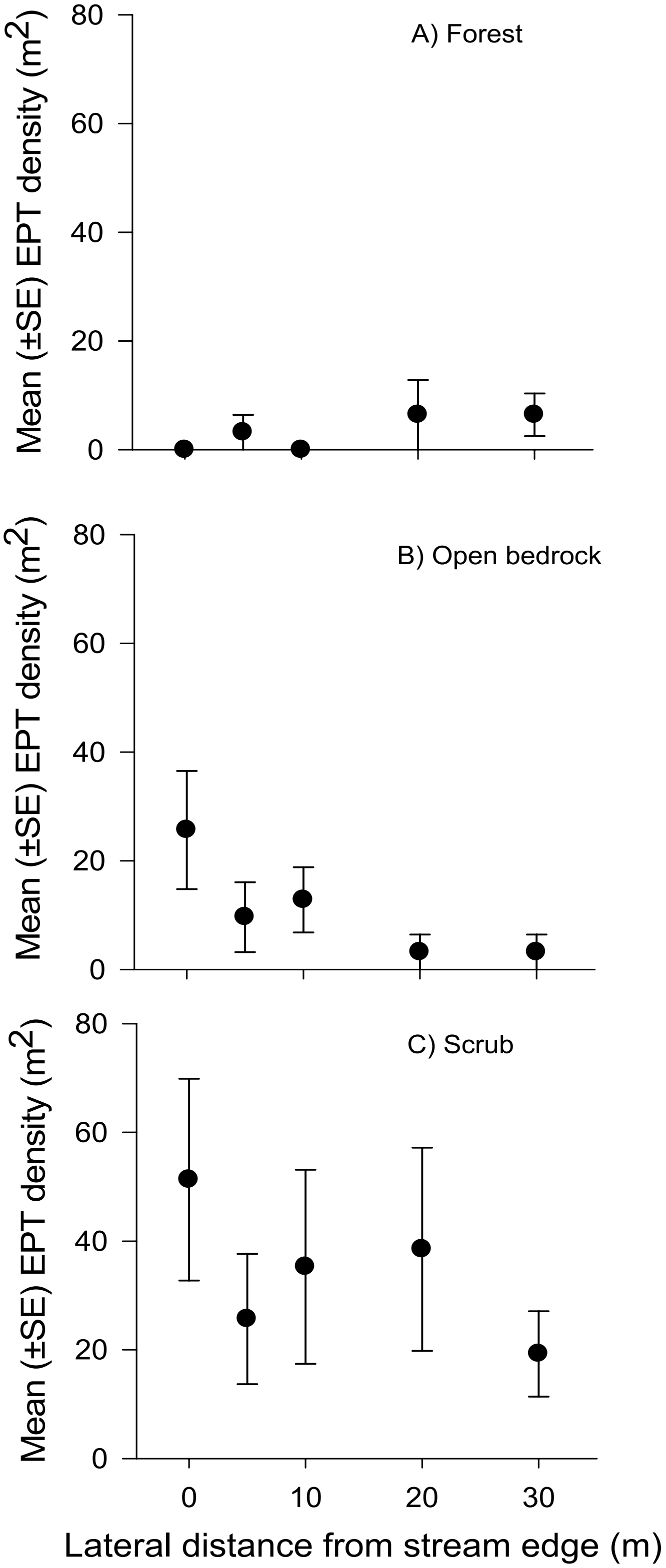


Figure 3.4: Mean (± 1SE) density (m^2) of adult aquatic insects caught on sticky traps at five distances from the T35 on the Stockton Plateau between November 2008 and January 2009.

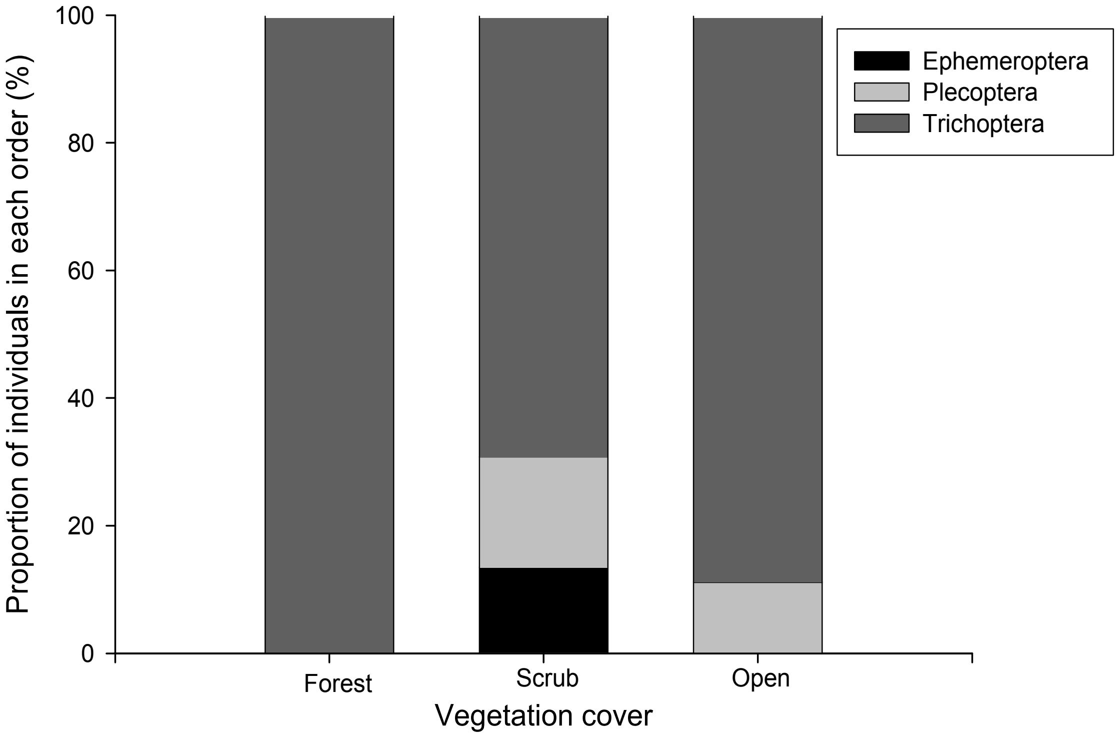


Figure 3.5: The proportion of individuals caught on sticky traps in each vegetation cover.

Downstream drift versus downstream flight

A one-way ANOVA showed that significantly more macroinvertebrates were drifting than were flying downstream ($F_{1,11} = 59.99$, $P < 0.001$) (Figure 3.6). Interestingly, no Ephemeroptera were caught in aerial flight, but they were the dominant taxa exported via drift. In contrast, the dominant taxa flying were Trichoptera which were the least dominant in the drift samples (Figure 3.7). Summary data in Appendix 2 revealed that the mayflies *Austroclima* sp., *Deleatidium* sp. and *Coloburiscus humeralis* as well as the caddisfly, *Oxyethira albiceps* were all prolific drifters relative to their benthic density. *Ameletopsis* sp. and *Zelandoptila* sp. were collected in drift samples but were both absent from the benthic samples. *Taraperla* sp. was particularly interesting given its relatively high benthic density yet was absent from the drift samples (Appendix 2).

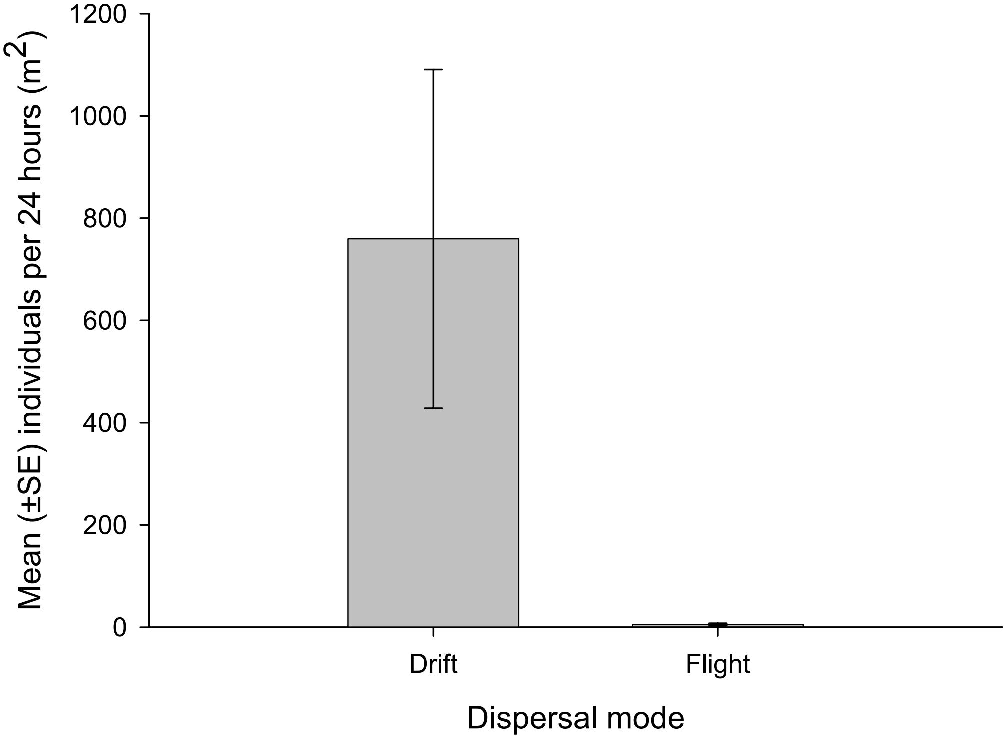


Figure 3.6: The mean (±SE) number of individuals caught in drift nets and sticky traps in a 24 hour period in six streams on the Stockton Plateau in a three day period in January 2009.

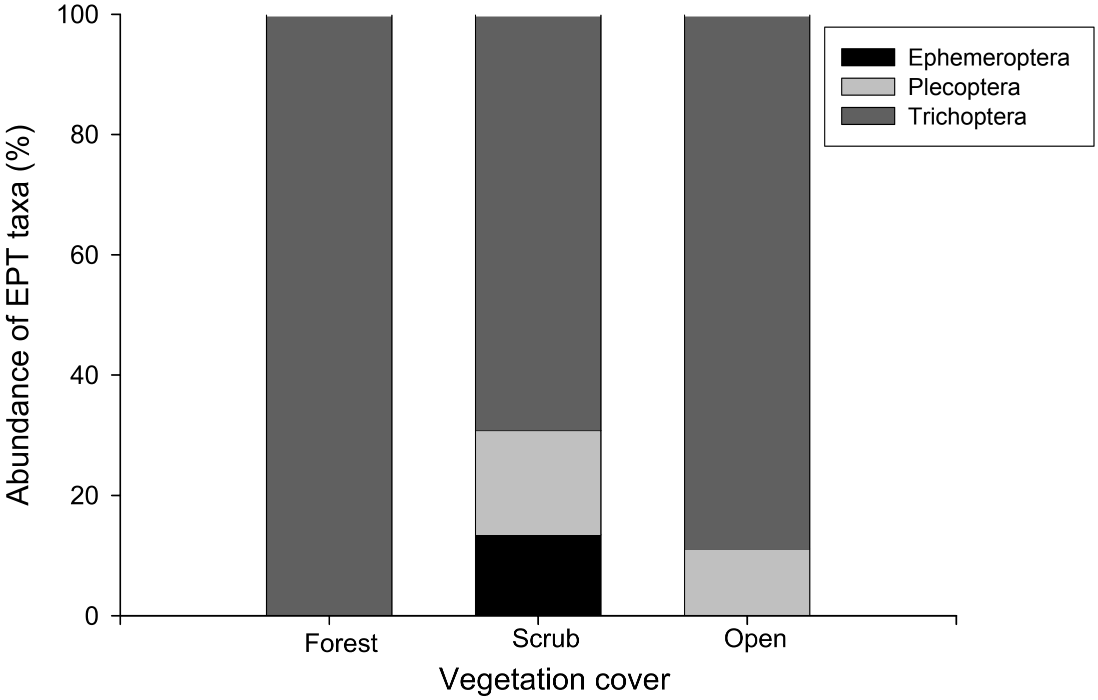


Figure 3.7: The proportion of EPT orders drifting and flying downstream in a three day period in six Stockton Plateau streams in January 2009.

Discussion

Aquatic adult insect flight and the influence of vegetation cover

Müller's (1954, 1982) colonisation cycle advocates that downstream movement of macroinvertebrates is compensated by predominant upstream flight of aquatic adult insects that lay their eggs further up-stream. Since this concept originated there have been a number of studies that have investigated this relationship and found a tendency for upstream flight (e.g. Bird and Hynes 1981, Griffith et al. 1998, Winterbourn and Crowe 2001, Collier and Quinn 2003, Elliott 2003, Winterbourn 2005). In contrast, my study did not detect any significant differences between upstream and downstream flight. This result is interesting as it might have been expected that upstream dispersal may be needed to compensate for catastrophic drift which is an artefact of the disturbance regime on the Stockton Plateau. An alternative theory to Müller's (1954, 1982) was proposed by Anhold (1995) who advocated that density-dependence and random dispersal by aquatic adult insects maintained stream macroinvertebrate populations as opposed to evolutionary or behavioural induced upstream flight. This theory has found support from Wagner (2003) who found no predominant upstream flight for any of the adult insects studied. However, I believe that it is unlikely Anhold's (1995) proposal applies to the Stockton Plateau streams given the complex topography and frequent disturbance. In Chapter Two it was identified that a streams geographic location within the plateau exerted a significant influence (albeit, of secondary importance to AMD) on macroinvertebrate community composition. This is potentially due to the streams being located in heavily incised gullies (Plate 3.5) which act as 'dispersal conduits' confining adult insects to the stream corridor. Consequently, random dispersal events seem less likely. Secondly, given the frequent flooding and the assumed resulting catastrophic drift, it is less plausible that density-dependence interactions will play an important role in these streams.

It is possible that the non significant result between upstream and downstream flight is an artefact of a combination of low trapping rates in my traps and the opportunistic emergence of aquatic adult insects that is common in New Zealand stream insects (Cowie 1980, Winterbourn et al. 1981, Hart 1985). Despite this opportunistic behaviour, it does not mean there is not a peak emergence; it is just as defined as it is in Northern

Hemisphere streams. Furthermore, in mid September there were more adult mayflies flying at some of the Stockton streams suggesting I may have missed the relative peak emergence of some adult taxa.

Plate 3.5: A heavily incised stream (MT6) which is characteristic of streams on the Stockton Plateau.

My study also considered the affect that different vegetation had on lateral dispersal distances. This study found that there was no significant effect of vegetation on lateral dispersal distances, nor was there a significant reduction in aquatic adult insect densities up to 30 m from the stream. These non significant results could also be partially the result of the low catch rates, opportunistic emergence of aquatic insects and missing peak emergence. Despite these results, general patterns were apparent. Notably, forested vegetation had very low numbers of aquatic insects dispersing into it, while scrub had the highest abundances at all lateral distances. In contrast, densities of adult aquatic insects started relatively high at 0 m at the open bedrock site, but rapidly declined with increasing distance. I had expected that vegetation type would significantly reduce aquatic adult densities with increasing distance from the stream, more so for forest than scrub and open bedrock. Despite my results, the patterns of lateral dispersal in this study are similar to studies from overseas and New Zealand. For example, the effect that vegetation had on chironomids in a French agricultural landscape was investigated by

Delettre and Morvan (2000) who quantified the lateral flight of chironomids through open farmland and dense hedgerows. They found there was an exponential decline of chironomid densities with increasing lateral distance from the stream where the decline in density was significantly greater in hedgerow vegetation. They suggested that this was because the dense hedgerows acted as an impenetrable barrier to the chironomids in contrast to the open farmland. Similarly, Collier and Smith (1998) found caddisfly abundance declined exponentially with increasing distance into riparian forest with the most rapid decline observed within 20 m. This was attributed to the forest restricting lateral dispersal due to the dense understory which was consistent with the conclusions made by Sode and Wiberg-Larsen (1993). In contrast, a Californian stream showed a number of taxa were abundant 150 m away from the stream with riparian vegetation that comprised a medium density of trees with large areas of open shrubland (Jackson and Resh 1989). Jackson and Resh's (1989) finding is important in the context of my study as the highest densities for each lateral distances for aquatic adult insects were recorded at the scrub site where there was a mixture of dense stands of tall manuka, sporadically located manuka and tussock which potentially provided enough shelter to avoid the influence of wind (e.g. Briers et al. 2003), but was not so dense that it prevented lateral flight (as was the potentially the case at the forest site).

The distances individuals disperse into riparian vegetation could be influenced by behavioural traits. Kovats et al. (1996) observed that there was a chaotic lateral distribution of *Cheumatopsyche speciosa* from a lake edge in Canada. They attributed this to females requiring stable resting places to allow for their eggs to mature after mating. In another lateral dispersal study of Plecoptera and Trichoptera, it was found that different species had differing preferences (Petersen et al. 1999). For example, *Leuctra fusca* and *Nemurella pictetii* were significantly more abundant and dispersed further in woodland than they did in open country were they were susceptible to wind.

Despite these direct trapping studies not being able to confidently conclude on the widespread lateral dispersal of adult insects between catchments, there are a number of studies that provide indirect evidence that small numbers of individuals disperse large distances to distant streams (e.g. Hughes et al. 1998, MacNeale et al. 2005a, Schultheis and Hughes 2005). These studies suggest there are a small number of individuals dispersing between catchments and are providing enough gene flow to homogenise populations and help maintain benthic.

The importance of downstream drift and aerial flight for the supply of re-colonists

The purpose of the comparison between macroinvertebrate drift and adult flight was to identify which mechanism might provide more colonists to a stream network that had been treated for AMD. It was not designed to contribute to the debate over the evidence and applicability of the drift paradox as proposed by Müller (1954, 1982). My comparison showed that there were significantly more individuals collected in drift nets than sticky traps. In particular, drift was dominated by Ephemeroptera in contrast to adult flight which was dominated by Trichoptera. These results suggest that drift is likely to provide more direct colonists to a downstream reach. In studies that have considered the methods of re-colonisation following disturbance (typically flooding) it has been found that drift provides the most immediate and dominate supply of colonists. In an artificially disturbed Swiss stream Matthaei (1996) found that drift was the most important method of re-colonisation. Furthermore, Gayraud et al. (2000) found supporting evidence that drift is the primary supplier of colonists to disturbed reaches. This conclusion was reached by comparing the colonisation of disturbed rock baskets that were placed in the substrate and sealed off in a way that only allowed individuals to colonise from the hyporheic. The remaining baskets were suspended and only allowed to be colonised from drifting macroinvertebrates (Gayraud et al. 2000). These findings suggest that drift is an important source of re-colonists in disturbed reaches. However, the importance of drift is influenced by the fact that there are patches within the stream that are un-impacted, thus providing individuals to drift and re-colonise disturbed reaches or streams. In contrast, streams that are undergoing remediation from AMD there may be no sources of colonists from within the stream. Therefore, drift can only act as a dominant source of colonists when there are intact headwaters or tributaries. If this is not the case, then re-colonisation will only occur via aerial dispersal. While my study collected very few dispersing adults the reproductive capacity of the adults is perhaps more important as one adult can lay hundreds of eggs. For example, at Repo stream *Coloburiscus humeralis* was one of the dominant mayflies and one female can lay between 3000 and 6000 eggs (Scarsbrook 2000 and references there-in).

Implications for stream remediation

My first aim was to investigate the influence of different vegetation on the lateral dispersal of aquatic insects. I found that different vegetation did not have a significant effect on lateral dispersal, nor did increasing distance from the stream edge. Nevertheless, scrub vegetation did seem to support higher densities of adult insects and at greater distances from the stream when compared to open bedrock and forest. The reason I selected these three vegetation types was because they are the dominant vegetation types on the Stockton Plateau. The open bedrock also mimics the barren overburden dump landscape typical of mine sites, while scrub is liable to occur on overburden dumps colonised by successional vegetation. The patterns I observed in this study suggest that scrub vegetation may facilitate greater dispersal of individuals further from the stream. However, one limitation of my research is that I only quantified lateral dispersal out to 30 m from the stream edge and I never was able to examine dispersal to, and the colonisation of nearby streams.

The comparison between drift and flight dispersal suggests that drift is likely to provide rapid re-colonisation to a downstream reach. Therefore, when selecting streams for remediation it is important that streams are selected that either have an intact headwater or un-impacted tributaries connecting the stream. Secondly, the patterns of the lateral flight suggest that scrub vegetation should be re-established on rehabilitated overburden dumps as quickly as possible so lateral dispersal between remediated and un-impacted streams can be facilitated.

Chapter Four

Remediating macroinvertebrate communities in streams post-mining: responses to habitat and organic matter additions

Introduction

In many cases the successful recovery of aquatic macroinvertebrate communities in streams treated for acid mine drainage (AMD) remains elusive (Wantanabe et al. 2000, Monteith et al. 2005, McClurg et al. 2007). One explanation is that mining companies rely on the "field of dreams hypothesis" to remediate degraded streams. The justification for this philosophy is that removing the original cause of the degradation will allow macroinvertebrate communities to recover naturally. However, while AMD is the cause of the initial degradation, the recovery of macroinvertebrate communities will be determined by community assembly rules, especially the supply of food resources and habitat availability (Palmer et al. 1997). Consideration of these factors is a pivotal step in improving the recovery success of macroinvertebrate communities (Lake et al. 2007).

Typically, riparian vegetation is removed during mining and as a result the amount of organic matter entering a stream is markedly reduced. This loss of organic matter might be expected to have profound bottom-up effects on stream macroinvertebrate and fish communities (Wallace et al. 1997, Wallace et al. 1999, Hall et al. 2000, Baer et al. 2001, McIntosh et al. 2005, Entrekin et al. 2007, Entrekin et al. 2008). Additionally, the loss of riparian vegetation can reduce terrestrial subsidies of terrestrial insects which can have a substantial top-down effect as predators switch from terrestrial to aquatic prey (Nakano et al. 1999, Greene et al. 2008, Huang et al. 2008, Leroux and Loreau 2008). In some anthropogenically impacted streams, restoring riparian vegetation or organic matter retention structures has resulted in improvements in the quantity of coarse particulate organic matter (CPOM) and macroinvertebrate density (Muotka and Laasonen 2002, Muotka et al. 2002, Negishi and Richardson 2003, Pretty and Dobson 2004, Muotka and Syrjanen 2007, Scealy et al. 2007, Lester and Boulton 2008). However, this is not a consistent response, and the macroinvertebrate community in a remediated urban stream failed to positively respond to enhanced riparian cover (Blakely and Harding 2005). Despite the acceptance that organic matter resources need to be improved to aid remediation, an assessment of the literature indicates little consideration has been given to the benefits that benthic communities by experience from the re-introduction of organic matter into streams treated for AMD.

The availability of habitat is also extremely important because macroinvertebrate recovery will be limited without adequate habitat (Lake et al. 2007). During mining, high sedimentation, changes in water chemistry and the loss of coarse woody debris (CWD) reduce the quality and availability of in-stream habitat (DeNicola and Stapleton 2002, Matthaei et al. 2006, Lester and Boulton 2008, Molinos and Donohue 2009). The result of this is usually an impoverished macroinvertebrate community (Harding and Boothroyd 2004, Cover et al. 2008). Habitat complexity is important in structuring macroinvertebrate communities because it provides refuge from predation (Schneider and Winemiller 2008) and floods (Death and Winterbourn 1994). Habitat complexity also fosters organic matter retention (Muotka and Laasonen 2002, Negishi and Richardson 2003) and provides stable substrate for autotrophic production (Death and Zimmermann 2005, Elenter et al. 2007). Increasing habitat complexity in degraded streams can lead to significant improvements in benthic communities (Johnson et al. 2003, Bond et al. 2006, Sarriquet et al. 2007, Scealy et al. 2007).

Most studies have been conducted in lowland streams impacted by agriculture or urban development, where CWD and other organic substrates have historically been removed (Erskine and Webb 2003). In contrast, in steep gradient, un-impacted bedrock streams, such as those on the Stockton Plateau in New Zealand, much of the natural habitat variability is provided by bryophytes. Mosses and liverworts can be important habitats for macroinvertebrates (Maurer and Brusven 1983, Brusven et al. 1990, Suren 1991, Korsu 2004) and can provide between three and four times more surface area than bedrock and support greater macroinvertebrate densities (Hutchens et al. 2004). Bryophytes also play an important role in retaining detritus and providing a stable substrate for periphyton to colonise, thereby providing additional habitat and food resources for macroinvertebrates (Suren 1990, Suren and Winterbourn 1992). Bryophytes are usually eliminated from streams that have been impacted by opencast mining through the effects of AMD (Stephenson et al. 1995). However, despite mosses and liverworts providing important habitat, I have been unable to find any published studies that have considered the benefits of restoring bryophytes in streams as macroinvertebrate habitat, yet alone in streams where bryophytes is the principle substrate. After streams have been treated for AMD and sedimentation has been managed, it can still take a long period of time for moss to naturally re-colonise. Therefore, despite post-mining improvements in water quality, macroinvertebrate communities may be very slow to recover due to limited habitat availability.

The first objective of this aspect of my study was to determine the response of macroinvertebrate communities to the addition of organic matter to streams. The second objective was to determine whether the addition of habitat in the form of artificial moss enhanced macroinvertebrate communities. Because there were no streams on the Stockton Plateau that had been treated for AMD at the time of this study, un-impacted streams were manipulated. If significant responses in macroinvertebrate communities were detected, it would suggest that organic matter and habitat additions would be beneficial in enhancing recovery of formerly AMD impacted streams.

I hypothesised that the added organic matter would be physically and biologically broken down and shredded and this material would subsequently be deposited on the benthic substrate. Macroinvertebrate communities would respond by showing increases in macroinvertebrate density, particularly of collector-browsers, shredders and filter-feeders due to enhanced organic matter resources. The addition of bryophyte habitat to streams should also increase macroinvertebrate density because of greater habitat availability.

Methods

Study Sites

The study was conducted in nine streams (Table 4.1) on the Stockton and Denniston Plateaus north of Westport, New Zealand. The Stockton Plateau ranges in elevation from 250 to 540 m a.s.l. and the Denniston Plateau ranges in elevation from 262 to 535 m a.s.l. Due to the rapid incline of the plateaus from the coast, both plateaus are subjected to orographic rainfall that exceeds 6,400 mm annually (Solid Energy unpublished data). Due to high rainfall and shallow soils, runoff is rapid. Streams are also subjected to rapid and sometimes sustained increases in discharge making them highly physically disturbed. The St Patricks Stream is the second largest river on the Stockton Plateau: in 2008 the annual mean flow was 5.3 m^{-3}, whereas the mean annual maximum flow was 55.9 m^{-3} (Solid Energy, Unpublished data). These streams have predominantly bedrock beds with patchy bryophyte cover. With the exception of T31 (100% cobbles), cobble substrates and boulders are only found in pools. Due to very hot summers, low winter air temperatures, shallow, infertile, acidic soils and a high number of fog days, forested vegetation is sparse and located only in gullies (Alarcon 1997). The dominant riparian vegetation at most sites was manuka (*Leptospermum scoparium*), tussock (*Chionochloa* spp.), and tangle fern (*Gleichenia dicarpa*). At six sites (Weka, MT6, MT6a, T69, Rapid, Control) there was no canopy cover over the streams. T35 and T31 had greater canopy cover (Table 1) as they were in more sheltered gullies.

Table 4.1: Summary of physical and chemical conditions of the manipulated streams on the Stockton and Denniston Plateaus sampled under base flow conditions on 30 September 2008.

Site	Substrate	Riparian vegetation	Canopy cover (%)	pH	EC ($\mu S_{25} cm^{-1}$)	DO (mg/L)	Width (m)	Depth (cm)
T69	Bedrock	Tussock	0	4.8	26	12.2	0.9-1.5	5.0-8.0
Denniston Road	Bedrock	Manuka, Broadleaf	50	4.3	27	10.3	1.5-2.8	1-2.5
MT6a	Bedrock	Tangle-fern Manuka	0	4.2	49	11.5	0.5-1.3	0.5-1.0
T31	Cobble	Manuka	20	3.2	272	12.1	2.5-3.2	6.0-10.0
Rapid	Bedrock	Manuka, Tussock, Tangle-fern	0	3.8	29	12.0	1.2-1.9	6.0-10.0
Weka	Bedrock	Manuka	5	4.1	53	11.8	1-2.8	20.0-80.0
Denniston Control	Cobble	-	0	4.2	35	10.8	0.5-0.8	2.0-7.0
T35	Bedrock	Manuka, Tangle-fern	10	4.3	20	12.5	0.8-1.4	1.0-3.0
MT6	Bedrock	Manuka, Tangle-fern	0	4.2	53	12.2	2.5-3.8	1.0-2.5

At all sites I manipulated the abundance of organic matter and habitat (i.e. added artificial moss) to determine if food and habitat availability limited macroinvertebrate communities: three stream sites received artificial moss habitat (in the form of astro turf tiles); three received wood and leaf additions; and three were control streams. I sampled the sites on three occasions: pre- (August 2008), mid- (November 2008), and post-experiment (January 2009).

Organic Matter Experimental Manipulations

Prior to the addition of leaves and wood, the macroinvertebrate community and existing organic matter concentrations in each treatment stream and three un-manipulated controls (T35, Denniston Control and MT6) were surveyed. Macroinvertebrate diversity and abundance were quantified by taking five Surber samples (0.04 m^2; 500 μm mesh) from randomly selected riffles and runs. The quantity of suspended fine particulate organic matter (FPOM) was estimated from three, one litre water samples. The frequency of natural woody debris and leaf packs (defined as any build up of leaves with a radius greater than 5 cm behind an object protruding from the water surface) was recorded along a 70 m reach.

Wood and leaves were added to three streams to determine whether they would increase invertebrate densities. Mixtures of leaves (*Nothofagus fusca* (Red Beech), *Acer* sp. (Maple), *Quercus* sp. (Oak)) were collected in Christchurch during May and June 2008, dried at room temperature or in an oven at $40^{o}C$ and stored until use. Three hundred onion bags (500 x 280 mm; 4 x 4 mm mesh size) were filled with 40 g (dry mass) of mixed leaves. In August 2008, 100 onion bags (Plate 4.1) were placed in each of three streams (Rapid 1, T31 and Weka Creek). Onion bags were arranged on five strings of 20 bags, each with one end anchored to the stream bank. Rocks or pegs were used to submerge the remaining onion bags. In addition, CWD was added to the same streams. In each of the three streams, 50- 1 m lengths of 5 x 10 cm rough-cut, untreated pine (*Pinus radiata)* were added (Plate 4.1). The pine timbers were tied together in pairs, submerged in the stream, and secured to the stream banks with PVC-coated wire rope.

Plate 4.1: An example of the onion bags and pine wood additions in Weka Creek as part of the organic matter experiment.

In early November 2008 a mid-experiment survey was conducted by taking five Surber samples up-stream and five Surber samples down-stream of the reach where the organic matter was added, thereby incorporating in-stream as well as between-stream comparisons. Surber samples were taken from randomly selected riffles and runs and the substrate was vigorously disturbed. Suspended FPOM samples were collected by taking three, 1 L bottles of stream water. The frequencies of naturally occurring CWD and leaf packs as well as additions of onion bag and CWD were also recorded along a 70 m reach. Macroinvertebrates were also removed from five pairs of pine timbers and preserved in 70 % ethanol. During the mid-experiment survey it was observed that many onion bags were deformed and had lost their leaves, probably because of physical abrasion during several high discharge events. As a result 50 onion bags with 40 g of untreated post peel (*P. radiata*) were added to each stream manipulated by organic matter in early December 2008. Remaining onion bags and pine timbers were removed, and organic matter and macroinvertebrates were re-surveyed in January 2009 using the same procedures as in the mid-term sampling.

In the laboratory, macroinvertebrate samples (Surber samples, onion bags and pine timbers) were washed over a 500 µm sieve and all macroinvertebrates were counted and identified to genus for Ephemeroptera, Plecoptera and Trichoptera taxa and family for all other orders using standard keys (Winterbourn et al. 2000b, Smith 2001). Macroinvertebrates were then categorised into functional feeding groups following Cowie (1983).Suspended FPOM was filtered from 1L water samples through pre-weighed 0.45 µ m Whatman GF/C. Filters were then dried at 50°C for 48 hours, weighed, ashed at 550°C for two hours, and weighed again. Ash free dry mass (AFDM) of FPOM was calculated as the difference between dry and ash masses (APHA 2005). CPOM per unit of stream bed was estimated from organic matter present in Surbers samples filtered through 500 µm sieve after all the macroinvertebrates had been removed. Material retained by the sieve was dried for 24 hours at 50°C and ashed in a pre-ashed, pre-weighed aluminium tray for four hours at 550°C.

Moss Experiment

Concrete paving tiles (400 x 400 x 30 mm, 5.5 kg) with astro turf (area 350 x 350 mm, strands 7 mm long, ~ 30 strands per cm^2) glued to the upper surface were added to streams to mimic moss covered bedrock substrate (Plate 4.2). Fifteen tiles were placed in each stream (MT6a, Denniston Road stream, and T69) in early August 2008. Prior to this addition, 5 Surber samples (0.04 m^2; 500 µm mesh) were collected from five randomly selected riffles and run, and the area of bryophyte cover was measured within seven 1 m^2 quadrats placed at 10 m intervals along a 70 m reach.

In early November 2008 a mid-experiment survey was conducted by collecting Surber samples (0.04 m^2; 500 µm mesh) from five randomly concrete tile moss additions and five randomly selected Surber samples from riffles and runs from the surrounding substrate. Macroinvertebrates were also collected from underneath the tiles by placing a kick-net (500 µm mesh) down stream of the tile before it was overturned and the macroinvertebrates washed off. As in August the area of bryophyte cover was measured in 1 m^2 quadrats every 10 m along a 70 m reach. The same procedures were followed for the post-experiment survey at the end of January 2009. Macroinvertebrates from both sampling periods were processed using the same method as for the organic matter addition experiment.

Plate 4.2: Artificial moss addition in T69. Astro turf simulating moss was glued to concrete paving tiles.

Statistical analysis

Repeated measures ANOVA was used to determine significant responses by physical and biological response variables to the resource additions between sampling periods using Statistica (Version 8) (StatSoft 2008). Biological response variables included total taxa, density, percent Ephemeroptera, Plecoptera and Trichoptera (EPT), percent dominance of functional feeding groups and the Berger-Parker index. This index provides a measure of importance by the dominant species where an increase in the index means a community has a higher taxonomic richness and a number of these taxa have a similar dominance (Berger and Parker 1970).

Student t-tests were also used to determine whether there were significant responses by macroinvertebrate community variables between manipulations and instream controls. As required, data was log transformed to meet the assumptions of normality and results were considered to be significant at $P < 0.05$. Detrended correspondence analysis (DCA) was also conducted on quantitative macroinvertebrate community data for control sites, manipulation sites and instream controls for each manipulation and using PC-ORD (McCune and Mefford 1999).

Results

Sampling of macroinvertebrates, benthic CPOM, moss cover, CWD and leaf packs prior to manipulations indicated there were no significant differences for any of the measured variables between streams (Table 4.2). All sites were dominated by collector-browsers (*Deleatidium*, *Austroclima*, *Zephlebia* and *Zelandoperla*) followed by predators which were dominated by *Megaleptoperla*, *Hydrobiosis* and *Psilochorema*. Shredders and filter-feeders had low proportional representation in the streams where they were found. Specialist grazing taxa were absent in all streams except Weka Creek (Figure 4.1).

Table 4.2: Mean (±SE) values of macroinvertebrate and habitat variables in a 70 m reach for three stream types (n =3 per stream type) on the Stockton and Denniston Plateaus in August 2008. Results of one-way ANOVAs comparing streams for the preliminary survey. (*F* and *P* vales are also provided. Significant values were accepted at $P < 0.05$.

Variable	Control	Organic matter addition sites	Habitat addition sites	df	*F*-value	*P*
Total taxa per site	17.3 (3.4)	8.33 (2.4)	10.3 (2.0)	2	3.08	0.12
Macroinvertebrate density (no.m^{-2})	575.6 (282.6)	255.0 (105.0)	311.3 (197.6)	2	0.50	0.63
CPOM (g m^{-2})	0.6 (0.2)	2.8 (1.4)		2	5.38	0.10
FPOM (gL^{-1})	0.08 (0.0001	0.09 (0.0004)	0.09 (0.0002)	2	2.63	0.15
% moss cover	57.4 (23.9)	20.65 (10.1)	40.6 (19.6)	2	0.35	0.72
CWD (cm^{2}.m^{-2})	1962.7 (1490.9)	610.6 (10.11)	442 (442)	2	0.79	0.5

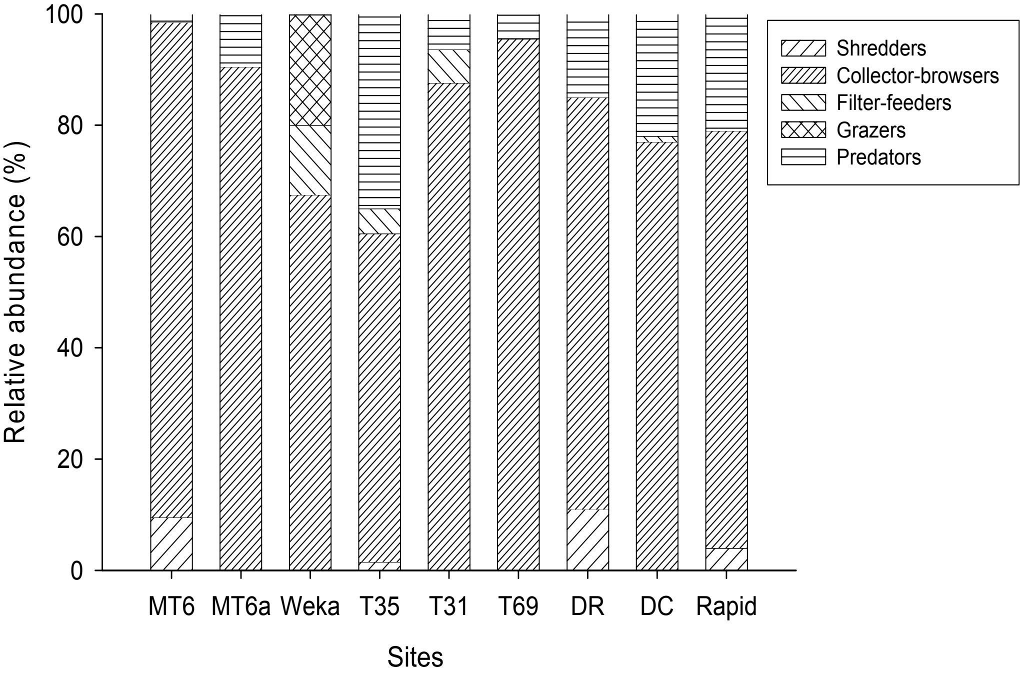


Figure 4.1: Relative abundance of functional feeding groups in 9 streams on the Stockton and Denniston Plateau for the preliminary survey in August 2008.

Organic matter additions

The additions of onion bags filled with leaves and CWD significantly increased the frequency of retained organic matter in the manipulated streams ($F_{1,5}$= 5404.6, P <0.01) (Figure 4.2). However, this significant increase in the frequency of organic matter did not correspond to an increase in benthic CPOM on the benthic substrate. Control sites had consistently lower quantities of benthic CPOM and concentrations were consistent throughout the addition (Figure 4.3). In contrast, manipulated sites showed a significant decrease between the pre- and mid-experiment sampling ($F_{2,\ 12}$ = 13.75, P = 0.02) (Figure 4.4b) before CPOM concentrations recovered to levels similar to those found in the preliminary survey. During organic matter additions there was a significant decrease in taxonomic richness between the preliminary and mid-experiment surveys (Table 4.3, Figure 4.4b). The proportion of collector-browsers in the macroinvertebrate community also significantly decreased between the preliminary and mid-experiment samples (Figure 4.4c, Table 4.3). Shredders and filter feeders had low densities in streams with organic matter additions and were not present in the mid and post-experiment surveys. Interestingly, the Berger-Parker index and the percent dominance of predators increased between the preliminary and mid-experiment surveys. When comparisons were made between macroinvertebrate community response variables for each manipulated reach and each in-stream control there were no significant differences between any of the variables (Table 4.4).

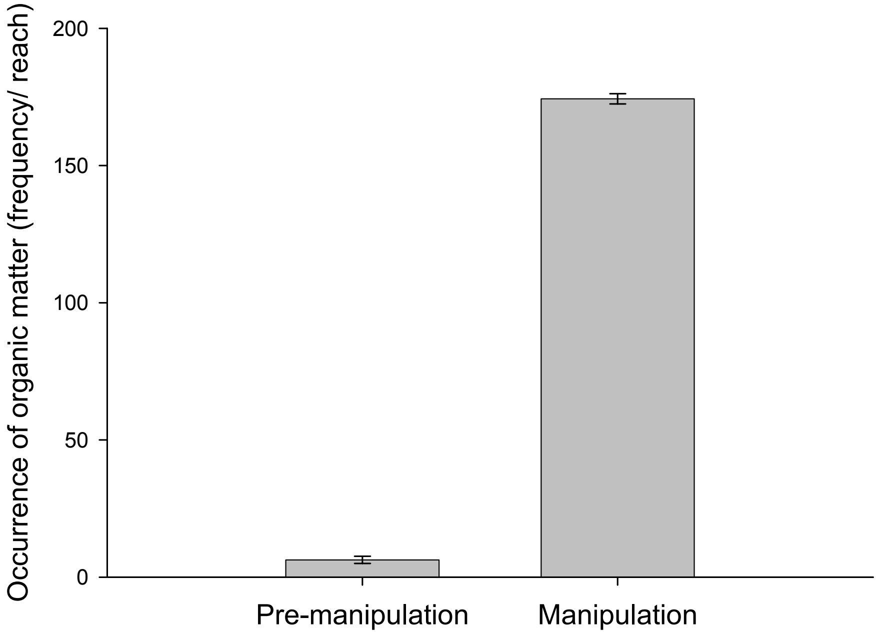


Figure 4.2: Mean (±SE) occurrence frequency of organic matter (CWD and leaf packs) in a 70 m reach for three streams on the Stockton and Denniston Plateaus with organic matter additions. Pre-manipulation sites are all the organic matter manipulation streams prior to the additions. Manipulation sites are the organic matter streams with the frequency of occurrence of CWD or leaf packs averaged between the mid-manipulation and post-manipulation surveys.

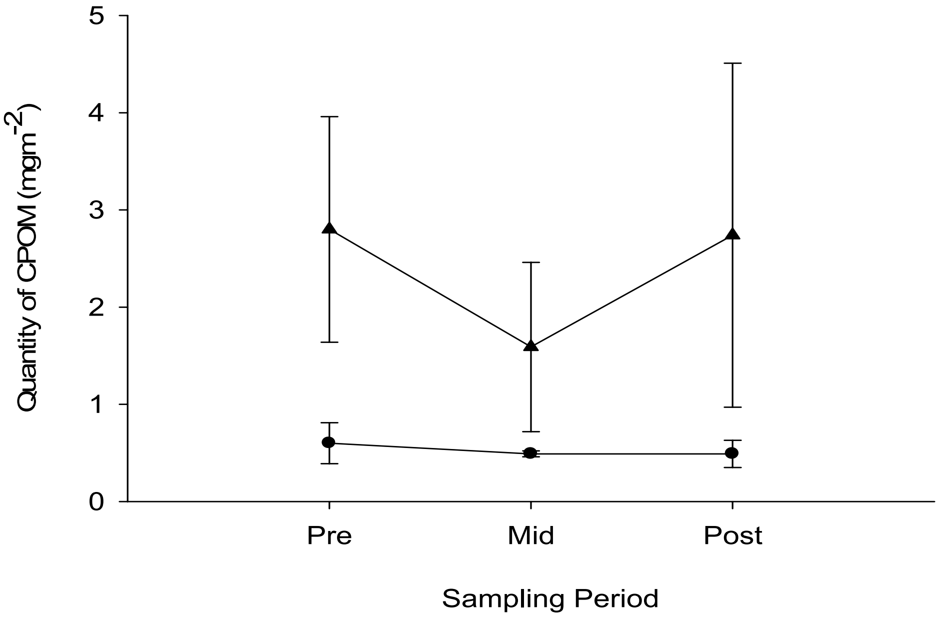


Figure 4.3: Mean (± SE) quantity of benthic CPOM in streams to which organic matter was added (closed triangles) (n =3) and control streams (closed circles)(n =3) on the Stockton and Denniston Plateaus in the three sampling periods between August 2008 and January 2009.

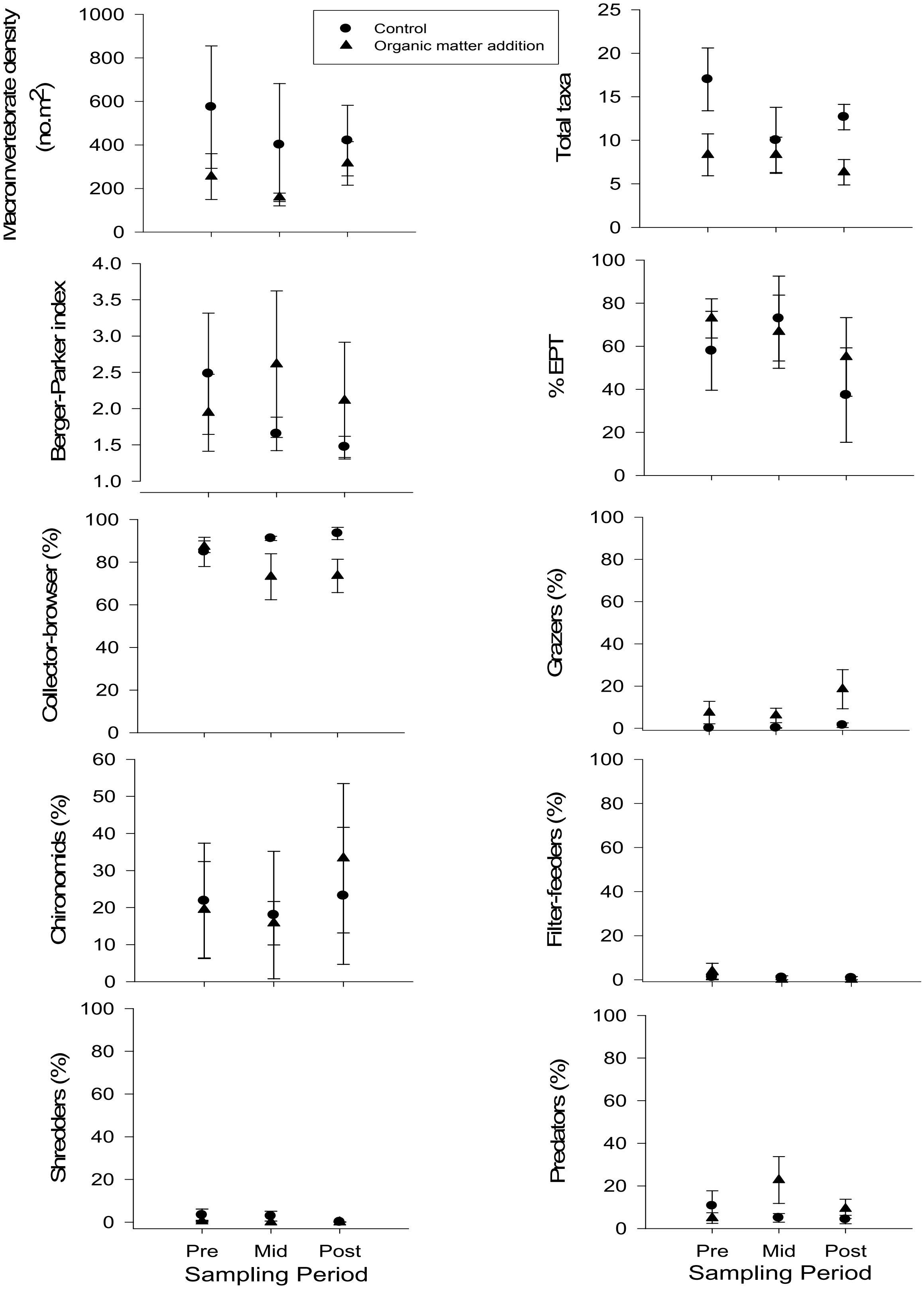


Figure 4.4: Mean (± SE) values of macroinvertebrate community response variables in the three sampling periods for nine streams on the Stockton and Denniston Plateau between August 2008 and January 2009.

Table 4.3: Results of repeated measures ANOVA comprising 9 macroinvertebrate variables for 6 streams (organic matter = 3, artificial moss = 3) sampled three times between August 2008 and January 2009. Responses were tested between manipulations (resource addition) and of manipulations (resource addition) over time. Significant values in bold.

Source of Variation	Error, df	F	*p*
Total taxa			
Resource addition	12, 2	2.13	0.20
Time	12, 2	4.27	**0.04**
Resource addition x time	12, 4	1.43	0.28
Macroinvertebrate density			
Resource addition	12, 2	0.99	0.43
Time	12, 2	0.59	0.57
Resource addition x time	12, 4	0.13	0.97
% EPT			
Resource addition	12, 2	2.07	0.17
Time	12, 2	2.14	0.20
Resource addition x time	12, 4	1.48	0.27
Berger Parker index			
Resource addition	12, 2	0.19	0.83
Time	12, 2	0.61	0.57
Resource addition x time	12, 4	0.47	0.76
% Collector-browsers			
Resource addition	12, 2	0.03	0.97
Time	12, 2	25.14	**0.01**
Resource addition x time	12, 4	0.91	0.49
% Predators			
Resource addition	12, 2	3.79	0.09
Time	12, 2	2.43	0.55
Resource addition x time	12, 4	2.43	0.10
% Shredders			
Resource addition	12, 2	2.62	0.11
Time	12, 2	0.89	0.46
Resource addition x time	12, 4	0.65	0.64
% Grazers			
Resource addition	12, 2	2.52	0.12
Time	12, 2	3.60	0.09
Resource addition x time	12, 4	0.29	0.88
% Filter-feeders			
Resource addition	12, 2	0.72	0.25
Time	12, 2	1.56	0.52
Resource addition x time	12, 4	0.78	0.56
% Chironomids			
Resource addition	12, 2	1.70	0.22
Time	12, 2	3.56	0.10
Resource addition x time	12, 4	1.95	0.17

DCA showed that the macroinvertebrate community for control sites T35 and Denniston Control (DC) were similar throughout the duration of the experiment. The final sampling period at T35 was an exception to this as it began to dry prior to the final sampling period and the bryophytes were exposed to the atmosphere (Figure 4.5). MT6 was distinct compared to the other control sites. The ordination also showed that the Rapid Creek and Weka Creek sites tended to cluster together, but there was no consistent trend for either of these sites on how the macroinvertebrate community responded to the organic matter additions. At site T31 the instream controls remain very similar between the mid and post experiment sampling. In contrast, the mid and post experiment communities were different from each other but followed the same path moving towards the middle of the ordination with Weka Creek.

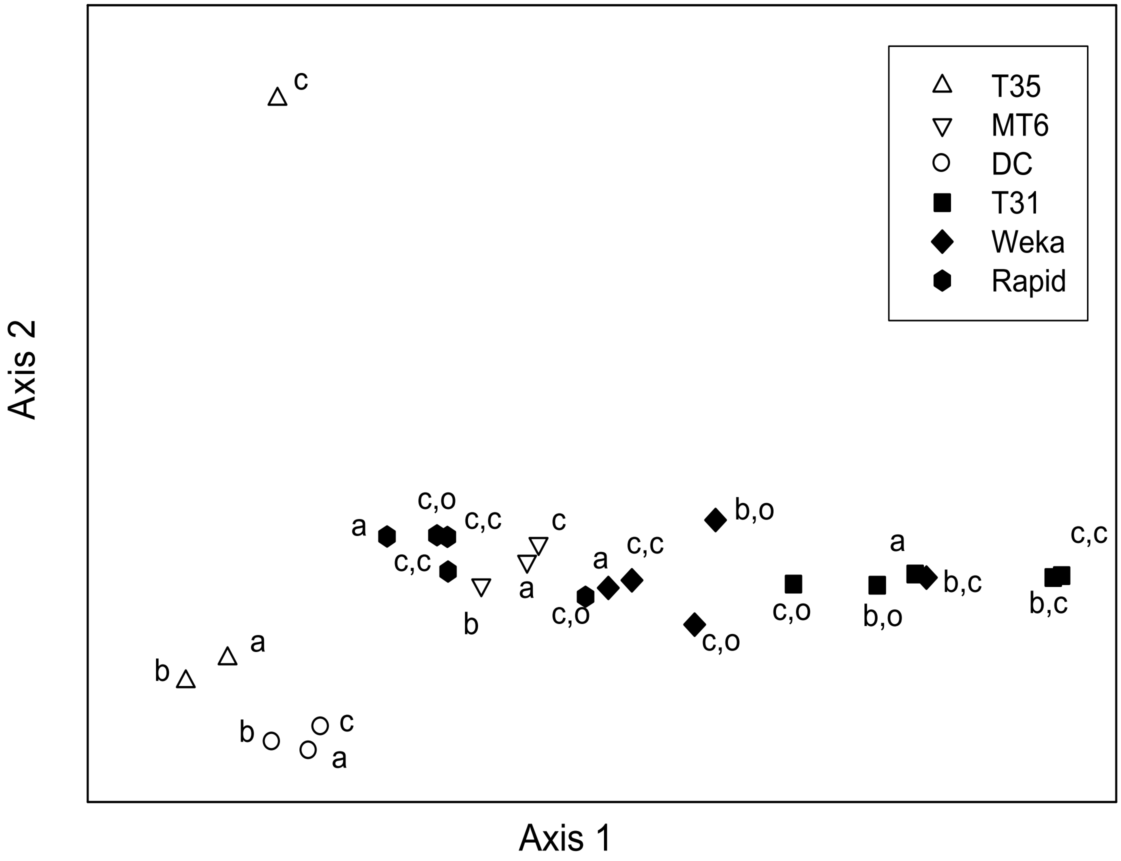


Figure 4.5: DCA plot showing community response to organic matter additions at three sampling times. Open symbols are control sites and closed symbols are addition sites. The first letter for each stream represents the sampling period (a = preliminary, b = mid, and c = post). The second letter is either ‘c’ meaning control or ‘o’ meaning organic matter additions. If there is only one letter it means these sites were control streams.

Table 4.4: Mean (± SE) values for macroinvertebrate community variables for three streams on the Stockton and Denniston Plateaus with organic matter manipulations and three up-stream controls between August 2008 and January 2009. Results of paired two-way t-tests comparing macroinvertebrate community response variables *F* and *P* values are also given and are significant at *P* <0.05.

	Manipulation	**Control**			
Response variable	**Mean (±SE)**	**Mean (±SE)**	**df**	**t-statistic**	***P***
Macroinvertebrate density	237.3 (57.3)	319.7 (107.5)	5	-0.34	0.7
Total taxa	7.3 (1.2)	7.5 (1.5)	5	0.09	0.9
% EPT	60.9 (11.5)	69.4 (13.0)	5	0.23	0.8
Berger-Parker index	2.4 (0.6)	1.8 (0.4)	5	0.69	0.5
% Chironomids	24.5 (10.2)	21.6 (13.1)	5	0.84	0.4
% Shredders	0 (0)	0.9 (0.9)	5	-1.00	0.4
% Collector-browsers	73.3 (5.9)	45.1 (14.4)	5	1.76	0.1
% Filter-feeders	0 (0)	0.9 (0.9)	5	-1.00	0.4
% Grazer	12.3 (5.2)	42.1 (17.6)	5	0.81	0.5
% Predators	16.0 (6.1)	11.9 (5.1)	5	0.30	0.8

Habitat (moss) additions

The addition of artificial moss substrate significantly increased the amount of moss (both natural and artificial) habitat in the manipulated streams ($F_{1,5} = 11.31$, $P = 0.02$) (Figure 4.6). However, two streams (T69 and Denniston Road (DR)) were highly susceptible to flooding and during each flood between three and five tiles were overturned in each stream. To compensate for this I moved the susceptible tiles to runs or pools. However, these tiles were then smothered by sediment. During the mid- and post- manipulation sampling I selected tiles that had not experienced sedimentation and were less susceptible to flooding throughout the manipulation. Of the macroinvertebrate community response variables measured, only taxonomic richness was significantly reduced between the preliminary and mid-experiment survey (Figure 4.7b, Table 4.3). Macroinvertebrate density followed a similar trend to control sites (Figure 4.7a) while the percentage of EPT taxa decreased between the preliminary and mid-experiment surveys (Figure 4.7d), the percent dominance of chironomids increased between the preliminary and mid-experiment samples (Figure 4.7g). The percent dominance of filter feeders, shredders and predators was low on artificial moss tiles when they were present (Figure 4.7h,i and j). When macroinvertebrate community response variables were compared between the artificial moss additions and the up-stream natural bryophyte control reaches there were no significant differences between any of the response variables (Table 4.5). The control sites (MT6, DC, and T35) maintained very similar communities throughout the study and were clustered together (Figure 4.8). T35 and DC were similar sites throughout the duration of the experiment while MT6 was distinctly different, but remain relatively constant during the experiment. The final sampling period at T35 was distinctly differed to the preliminary and mid-experiment samples as it began to dry prior to the final sampling period and the bryophytes were exposed to the atmosphere. The habitat manipulation sites were clustered together at the top left of the ordination graph, however, each site showed no consistent trend between all the sampling periods.

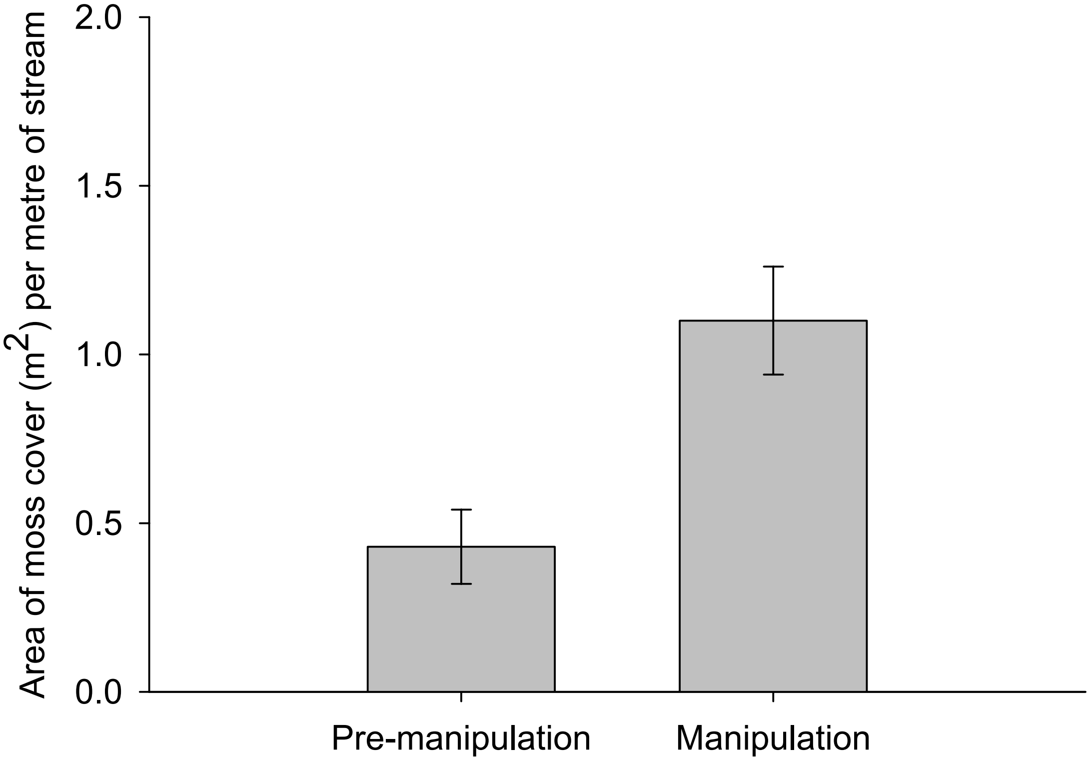


Figure 4.6: The area of moss cover ($m^2. m^{-1}$) on the wetted stream bed of moss-addition streams before the addition of artificial moss tiles (pre-manipulation) and during and after the manipulation (measured after three and six months combined and averaged).

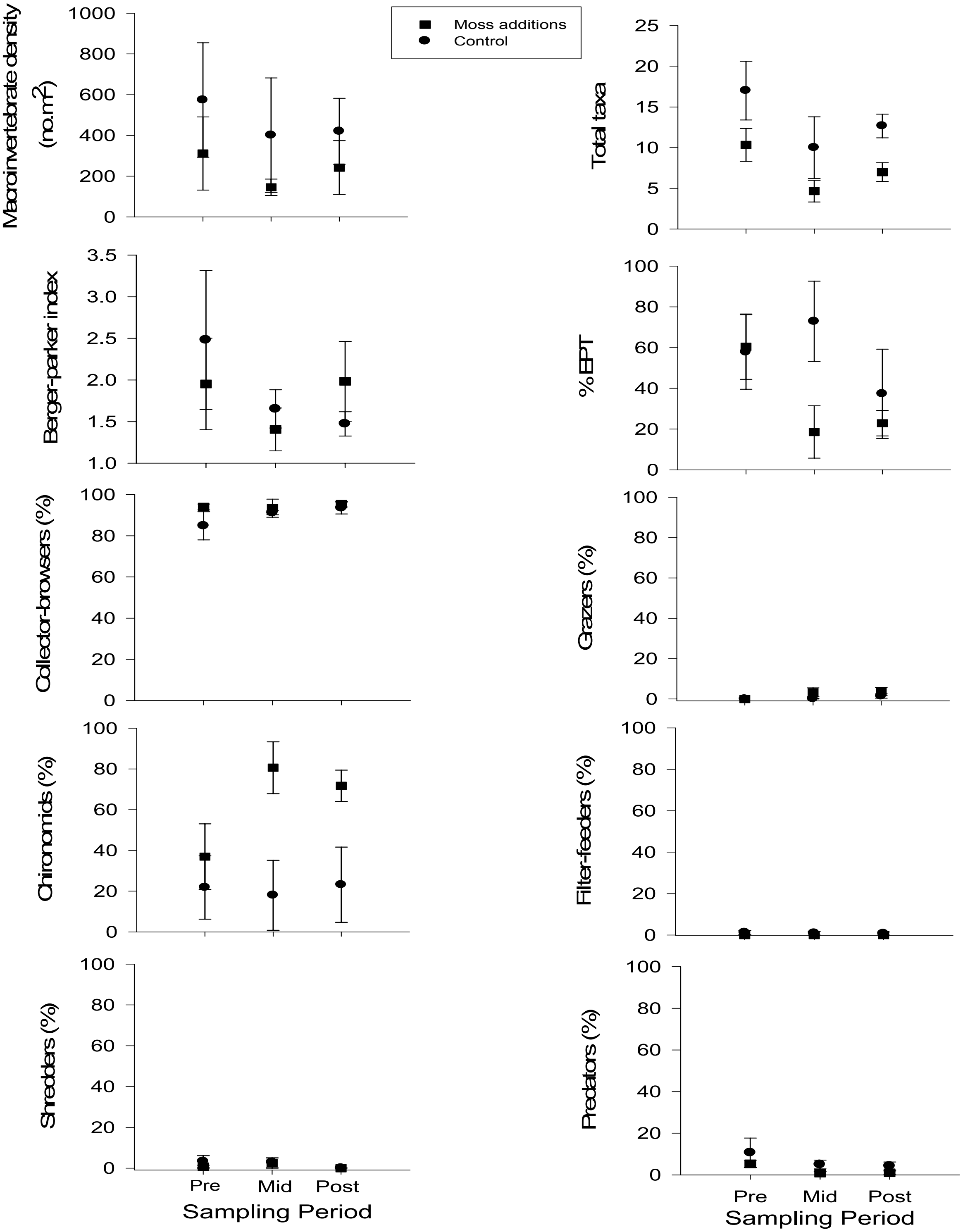


Figure 4.7: Mean (± SE) values of macroinvertebrate community response variables in the three sampling periods for nine streams on the Stockton and Denniston Plateau between August 2008 and January 2009.

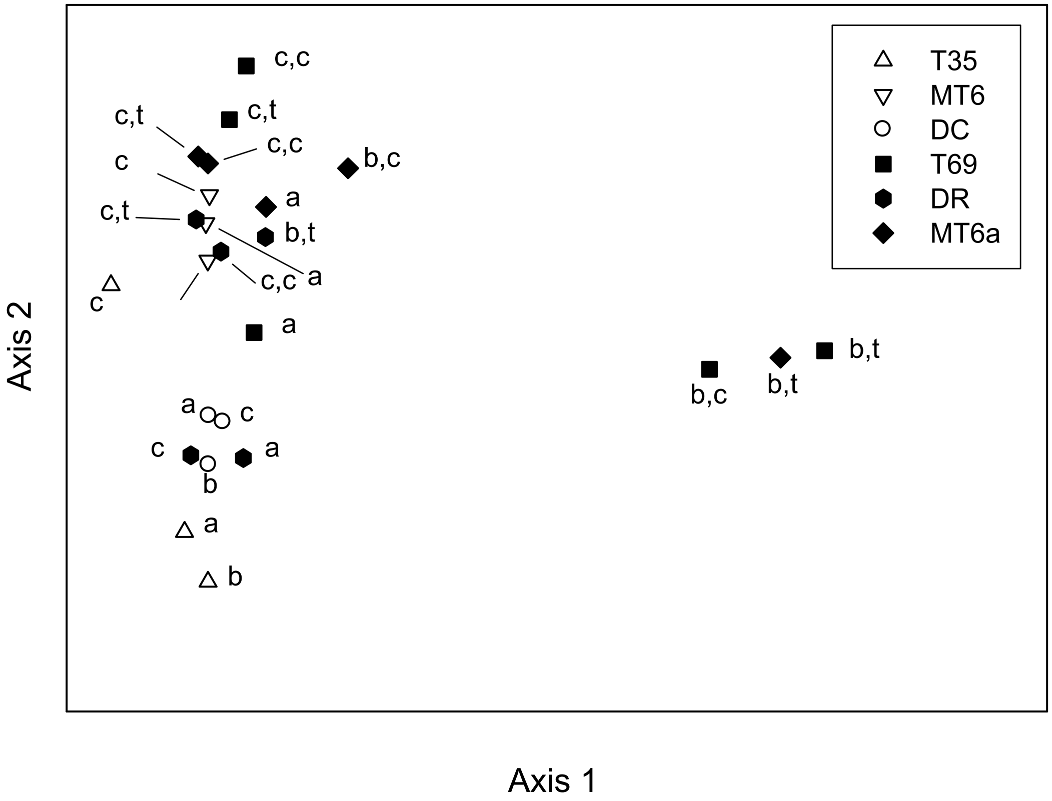


Figure 4.8: DCA plot showing community response to artificial additions at three sampling times. Open symbols are control sites and closed symbols are addition sites. The first letter for each stream represents the sampling period (a = preliminary, b = mid, and c = post). The second letter is either ‘c’ meaning control or‘t’ meaning artificial moss additions. If there is only one letter it means these sites were control streams.

Comparisons between the density and composition of macroinvertebrates underneath the tiles, on top of tiles and on the control benthic substrate showed there were significantly more predators beneath tiles ($F_{2,6} = 9.55$, $P < 0.001$) (Figure 4.9a) but proportions of other functional groups did not differ between habitat types. Significantly more crayfish (*Paranephrops zealandicus*) and mayflies (*Zephlebia* and *Deleatidium*) were found underneath tiles than on top of the tiles or in natural bryophyte habitat (Figure 4.9b). In contrast, CWD additions (pine timbers) had significantly fewer individuals on them than the adjacent benthic substrate (Figure 4.10).

Table 4.5: Mean (± SE) values for macroinvertebrate community variables for three streams on the Stockton and Denniston Plateaus with artificial moss additions and three up-stream controls between August 2008 and January 2009. Results of paired two-way t-tests comparing macroinvertebrate community response variables *F* and *P* values are also given and are significant at *P* <0.05.

	Manipulation	**Control**			
Response variable	**Mean (±SE)**	**Mean (±SE)**	**df**	**t-statistic**	***P***
Macroinvertebrate density	193.7 (65.4)	182.5 (49.8)	5	-0.08	0.94
Total taxa	5.83 (1.0)	9.0 (1.6)	5	-1.68	0.15
% EPT	20.7 (6.5)	44.0 (11.4)	5	-1.27	0.26
Berger-Parker index	1.7 (0.3)	2.1 (0.3)	5	-0.76	0.48
% Chironomids	76.2 (6.9)	46.3 (13.1)	5	1.36	0.23
% Shredders	1.2 (0.03)	1.1 (1.0)	5	0.03	0.97
% Collector-browsers	94.34 (2.1)	76.8 (10.5)	5	1.40	0.22
% Filter-feeders	0 (0)	2.3 (2.3)	5	-1.00	0.36
% Grazer	3.5 (1.3)	12.3 (10.7)	5	0.25	0.81
% Predators	0.9 (0.5)	7.5 (2.6)	5	2.08	0.09

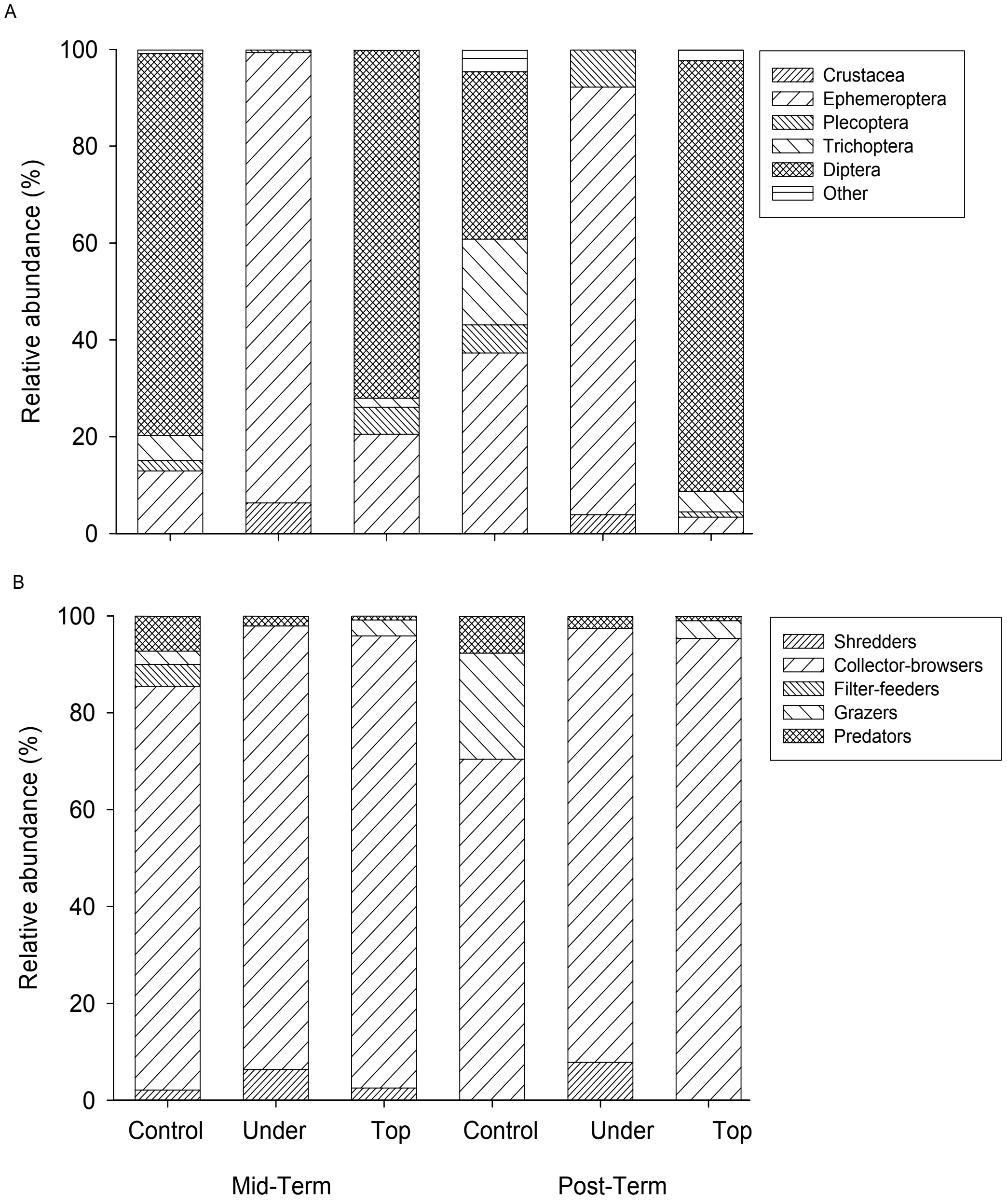


Figure 4.9: Relative abundance (%) of macroinvertebrate orders (a) and functional feeding groups (b) on top of artificial moss tiles, under tiles and on natural bryophyte cover in the mid- and post-term sampling periods in the three streams that had artificial moss additions during August 2008- January 2009.

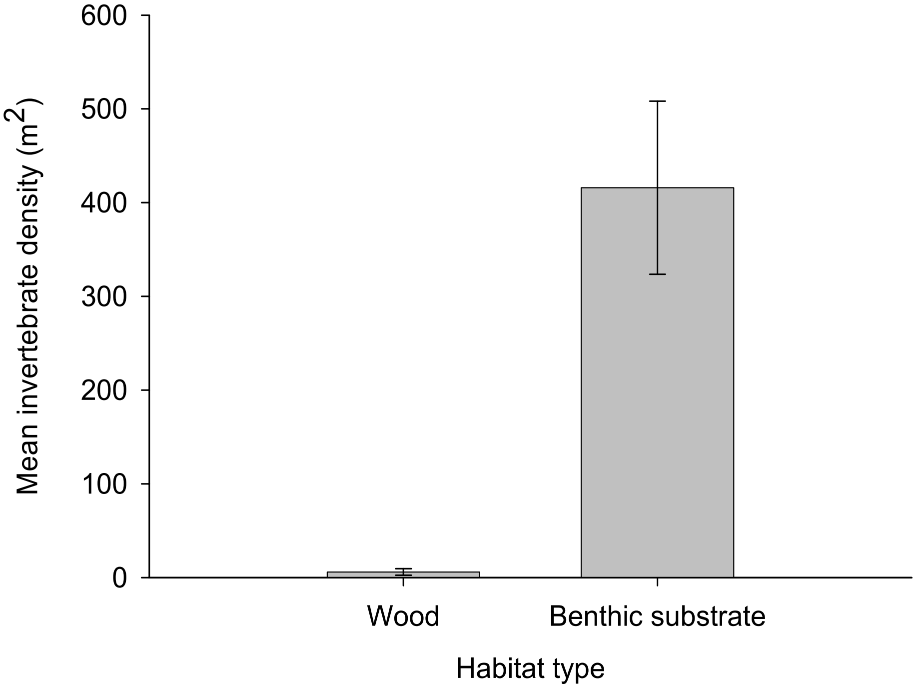


Figure 4.10: Mean (±SE) macroinvertebrate density on CWD additions and benthic substrates in three streams on the Stockton and Denniston Plateau between August 2008 and January 2009.

Discussion

Organic matter manipulations

Onion bags filled with leaves in addition to CWD were added to three streams to enhance concentrations of CPOM on benthic substrate as it was assumed physical and biological breakdown of the additional organic matter would deposit CPOM on the benthic substrate. In response to the hypothesised increase in CPOM on the benthic substrate, it was expected that macroinvertebrate density would increase for collector-browsers, shredders and filter-feeders. However, despite significantly increasing the occurrence of CWD and leaf packs in three streams, there was no significant increase in benthic CPOM. The only two macroinvertebrate community response variables to display a significant change were taxonomic richness and percent collector-browsers, but these were in response to declines between the preliminary and mid-experiment surveys. The non-significant change in the quantity of benthic CPOM and the significant reductions in percent collector-browsers (which dominated the stream communities) and total taxa, were likely the result of flooding. During the six month period of my study, over 3300 mm of rain fell, primarily during seven major storms; on each of these occasions >100 mm of rain fell in 24 hours. Two thirds of this rain fell between the preliminary sample and mid-experiment sample, including over 650 mm in the two weeks before the mid-experiment sample (Plate 4.3). The consequence of this flooding was probably the export of CPOM that had built up from the organic matter additions during base flow. Such an effect was shown by a numerical model whereby in a restored channel with five times more retentive structures than natural streams, an additional 15 % of CPOM was retained (Small et al. 2008). However, after six successive floods, the remaining organic matter was less than 50 % of that retained in the natural reach. This low retention was attributed to poor retentive capabilities of the structures in the restored reach (Small et al. 2008). Similarly, in a lowland stream in Arizona, Compson et al. (2009) reported flood discharges reduced leaf retention despite adequate trapping structures, as did Quinn et al. (2007). As studies have shown that the retention of organic matter is important for macroinvertebrate production, particularly so in forested streams (Wallace et al. 1997, Wallace et al. 1999, Bis and Higler 2001, Milner and Gloyne-Philips 2005, Aldridge et al. 2009), the loss of this material can have pronounced bottom-up effects on the food chain (Wallace et al. 1997, Wallace et al. 1999, Bis and Higler 2001, Milner and Gloyne-Philips 2005, Aldridge et al. 2009). However, in New Zealand mountain streams, organic matter is limited due to frequent flooding (Winterbourn 1982, Thompson and Townsend 2000) which was apparent in my manipulation streams. As a result, macroinvertebrates were not able to exploit this resource as would have been expected in less disturbed streams (Linklater and Winterbourn 1993, Harding and Winterbourn 1995, Collier and Smith 2003).

Despite my organic additions not providing an enhanced food resource due to flooding, it is possible that in these systems CPOM is less important than FPOM and biofilms. For example, Armstrong (1996) found that *Deleatidium* nymphs in 12 streams with limited canopy cover primarily consumed detritus, in contrast to Rounick and Winterbourn (1983) who found that 74 % of *Deleatidium* gut contents comprised algae. In a brown water stream near Reefton, it was found that *Deleatidum* gut contents primarily comprised FPOM (Cowie 1980). These studies suggest that the feeding strategies utilised by *Deleatidium* and other collector browsers allow such taxa to switch between food resources depending on their availability. Thus, in my experimental manipulation streams, it is possible that the dominant collector-browsers were consuming food resources other than what I sampled such as biofilms or FPOM retained in bryophytes. (Cowie 1980, Winterbourn 1982, Collier and Winterbourn 1990, Armstrong 1996). While the literature cited above is from pristine sites, it still suggests that reintroducing biofilms into AMD treated streams will potentially benefit macroinvertebrate communities as will appropriate habitat (bryophytes) where FPOM can be retained and biofilms such as diatoms can colonise (Rounick and Winterbourn 1983, Suren 1991, Suren and Winterbourn 1991, 1992, Wallace et al. 1999).

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a) b)

Plate 4.3: Rapid Creek study site, a) under baseflow conditions; b) following approximately 120 mm of rain in 24 hours. The photo on the right was taken one week before the mid-experiment sampling. The same large rocks have been circled.

Habitat (moss) additions

Artificial moss was added to three streams to determine if moss habitat was an important resource to add to streams remediated from AMD in order to facilitate macroinvertebrate recovery. Additions of artificial moss habitat significantly increased the availability of moss habitat in the manipulated streams. In moss manipulated streams, taxonomic richness was the only macroinvertebrate community variable to show a significant change between the preliminary and mid-term samples, and this was most probably the result of a decline after flooding that occurred during this period. The rapid colonisation of artificial moss by taxa representative of natural substrates and non-significant differences for all macroinvertebrate community response variables between the natural bryophytes and artificial moss indicate that artificial moss additions provided an appropriate habitat. Therefore, to help stimulate the recovery of macroinvertebrate communities, bryophyte cover must be reintroduced into these streams after AMD has been treated. This is vital because bryophytes are the dominant habitat on the Stockton Plateau and the rapid recovery of bryophytes will make macroinvertebrates less susceptible to flooding (Suren 1991, Suren and Winterbourn 1992, Milisa et al. 2006). In one of the limited studies of bryophytes in New Zealand, Suren (1991) found that water velocities were lower in moss than in inorganic substrate, which led to significantly higher macroinvertebrate densities. It was also found that bryophytes retain more organic matter (272.4 g.m^{-2}- 335.6 g.m^{-2}) compared to inorganic substrate, which only retained between 59.1 and 77.9 g.m^{-2} (Suren 1991). This indicates that in New Zealand, bryophytes not only provide valuable habitat, but they also provide efficient retentive structures for organic matter (allochthonous and autochthonous) in flood disturbed streams. Therefore, it is probable that the remediation of bryophytes of the Stockton Plateau will improve habitat for macroinvertebrates and protect them from the frequent flood flows as well as provide retentive abilities for FPOM and stable substrate for biofilm. However, no New Zealand study has considered the active remediation of bryophytes and there are no successful examples internationally (Reeves et al. 2004).

Another interesting observation was significantly more crayfish and mayflies were found underneath tiles than on control substrates and in the artificial moss. Crayfish are known to favour cover, such as weed beds, the undersides of stones, logs and overhanging banks (Winterbourn, 2004). As this habitat is limited in many Stockton Plateau streams, the addition of the concrete tiles appears to have benefited crayfish because it created additional stable habitat. Furthermore, when the tiles were turned over during my sampling it was noted that twigs and leaves were able to build up on the up-stream side of the tiles (retaining organic debris that was not normally abundant). As organic matter can be an important part of crayfish diets (Winterbourn 2004), the retention of this material by tiles may have increased the availability of food for them. The significantly greater abundance of mayflies under the tiles was possibly due to photo-negative behaviour whereby individuals are avoiding light (Hughes 1966, Horner and Goldman 1994).

The CWD addition was originally intended to provide a source of food for macroinvertebrates. However, it was quickly recognised that macroinvertebrates, in particular mayflies (*Nesameletus, Zephlebia* and *Ameletopsis*) that colonised CWD, did not feed on them. Significantly fewer macroinvertebrates were found on CWD additions than benthic substrate. Nevertheless, the fact that common macroinvertebrates colonised CWD is important for management of mine-impacted streams that have experienced significant sedimentation as the re-introduction of CWD can enhance the habitat available for macroinvertebrates as demonstrated by a number of studies (e.g. Benke et al. 1985, Collier and Halliday 2000, Collier et al. 2004, Bond et al. 2006, Lester et al. 2007).

Management implications

Despite significantly enhancing the abundance CWD and organic matter, there was no corresponding increase in benthic CPOM, which is most likely an artefact of flood disturbance. Flood disturbance limits CPOM retention and the size and potentially the composition of the macroinvertebrate community. Therefore, when remediating mine-impacted landscapes in areas of high disturbance, the remediation of riparian vegetation will provide some benefits (such as bank stabilisation, improved temperature, shading and dissolved oxygen parameters, as well as providing terrestrial subsidies such as insects and leave matter) (Quinn et al. 1994, Nakano et al. 1999, Wallace et al. 1999, Lester and Boulton 2008). However, it is unlikely that terrestrial subsidies of CPOM will correspond to an enhanced macroinvertebrate community. Therefore, it is apparent that the only food resource that collector-browsers can exploit in these highly disturbed and poorly retentive systems is biofilm and FPOM that is retained in bryophytes. Consequently, macroinvertebrate recovery is likely to be assisted by reintroducing diverse periphyton communities and FPOM.

CWD additions did provide an additional source of habitat for macroinvertebrates (albeit significantly less important than benthic substrate). However, this was unstable (one end was tied to riparian vegetation and CWD floated in the water), and during the next flood all the macroinvertebrates were washed away. The introduction of solidly embedded CWD will provide a stable substrate as suggested by Collier and Haliday (2000), Johnston et al. (2003) and Lester et al. (2007), as well as providing a trophic subsidy as periphyton colonises the CWD (Bond et al. 2006). Additions of CWD to streams that have experienced pronounced sedimentation will be important as this will provide stable substrate that can be used as habitat and for CPOM retention and biofilm colonisation.

Artificial moss additions did provide habitat for macroinvertebrates, and therefore, the reintroduction of this habitat should be encouraged. Re-introducing this habitat will also be important as a substrate for biofilm colonisation and FPOM retention, providing energy for macroinvertebrate consumption. A method to restore moss cover still needs to be addressed. Importantly, improving knowledge of moss life cycles and growth rates would be beneficial as will methods of how to reintroduce moss. Potential methods are identified in Chapter 5.

Chapter Five

Synthesis

Background

Evolving perceptions about the way humans interact with their natural surroundings have resulted in fundamental changes to environmental legislation (Memon and Perkins 2000), which have had marked effects on mining companies operations. In New Zealand, environmental legislation is embodied in the Resource Management Act (1991). One of the overriding principles of the Act is to avoid, mitigate or remediate any negative environmental effects (MfE 2006). However opencast mining is by its very nature a destructive process and it is often impossible to 'avoid' and difficult to 'mitigate' negative effects. For this reason, it is common practise to focus on restoring and remediating mine sites after mining has ceased. Both the type of ecosystems affected and mining processes can strongly influence the remediation method adopted at a mine. Depending on the presence of pyrite in the surrounding mine site, streams draining such mine sites can be effected by acid mine drainage (AMD) which has significant adverse effects on freshwater ecosystems (Harding and Boothroyd, 2004). While the effects of AMD on macroinvertebrate communities have been well documented and are relatively well understood (Courtney and Clements 2000, Winterbourn et al. 2000a, Courtney and Clements 2002, DeNicola and Stapleton 2002, Harding and Boothroyd 2004, Gerhardt et al. 2005) and the methods of treating AMD are well advanced and effective (O'Sullivan 2005), there have not always been improvements in stream communities in general, and macroinvertebrates in particular (Wantanabe et al. 2000, Monteith et al. 2005, McClurg et al. 2007).

This failure to stimulate macroinvertebrate community recovery is often the result of a failure to consider the effect community assembly rules have on the remediation of macroinvertebrate communities. Assembly rules are filters that operate at different spatial scales and structure a community by only allowing individuals that are capable of passing the filters to colonise the new system. Filters include regional processes such as constraints on dispersal, the regional species pool, and local processes such as habitat requirements and biotic interactions. Both regional and local filters structure the community by restricting colonisation of organisms that are unable to pass through the filter (Figure 5.1). This could be because the species has been eliminated from the regional species pool or because it is unable to disperse into the system, or is unable to compete with other organisms. Traditional remediation efforts however, have generally relied on the field of dreams hypothesis whereby “if you fix it they will come” Palmer et al. (1997).

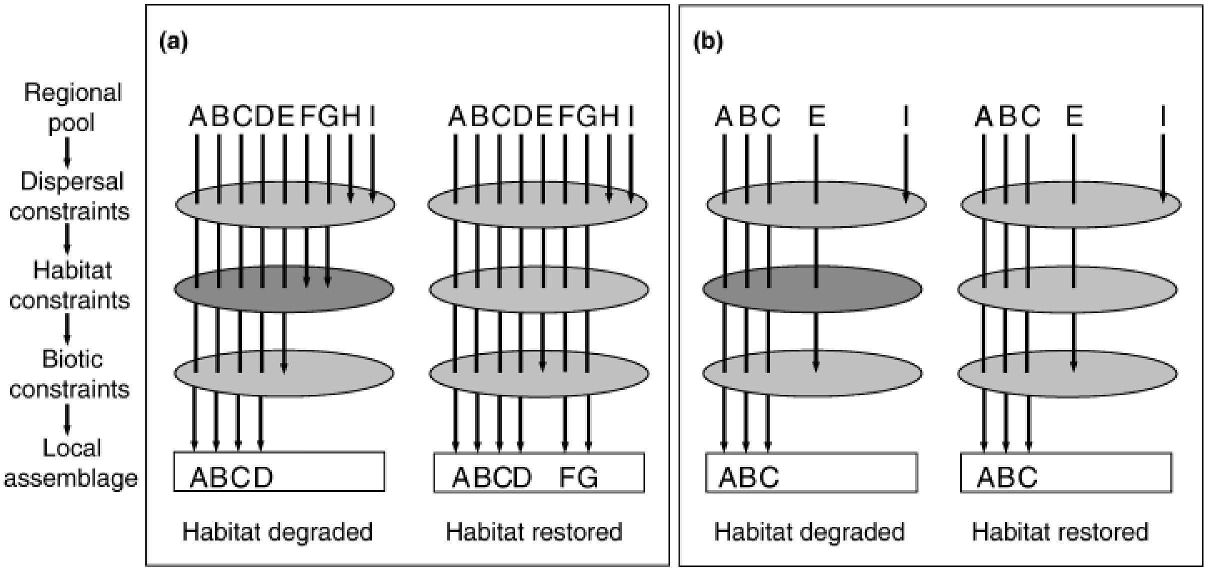

Figure 5.1: A theoretical model depicting the influence of regional and local scale filters on community composition. (a) shows the influence of the filters when the species pool is intact; (b) shows the influence of the filters when taxa have been eliminated from the species pool (Lake et al. 2007).

The last decade has seen significant advances in our understanding of the links between ecological theory and remediation ecology. For example, Palmer et al. (1997) advanced the idea that a remediated community is a function of both regional processes and local processes. This function is the relationship between filters that operate at both regional and local scales and which have been described above and in Figure 1. Other key concepts were identified and discussed by Lake et al. (2007) who emphasised the importance of considering and incorporating ecosystem processes that aid in structuring communities such as disturbance histories, connectivity between floodplains, the riparian zone and ground water as well as food web theory and succession. Consideration of these issues should enable practitioners to improve success rates in remediating macroinvertebrate communities in streams that have been treated for AMD because all factors that control the structure of a community have been re-introduced (Lake et al. 2007). However, at present, mining companies and their advisors do not consider community assembly rules in their remediation strategies and simply rely on the field of dreams hypothesis to achieve community recovery.

Key findings from my research

In this chapter I review the importance of some of the ecological theories proposed by Palmer et al. (1997) and Lake et al. (2007) in relation to the remediation of macroinvertebrate communities in streams treated for AMD. Specifically, I consider the locality of source populations of macroinvertebrates and the macroinvertebrates present that will constitute remediated macroinvertebrate communities. Secondly, I consider dispersal constraints on adult aquatic insects and their primary mode of re-colonisation. Finally, I consider food web theory and habitat requirements of macroinvertebrates. To do this I have examined: 1) how the position of streams within a landscape, (in conjunction with an AMD gradient) influences macroinvertebrate community composition. This has also been used to identify the regional pool of species that may be potential sources of colonists for future remediation projects; 2) the most likely mode of macroinvertebrate dispersal and the effects that vegetation has on the lateral dispersal of adult aquatic insects for re-colonisation; and 3) the effect of enhancing potential food and habitat (organic matter and moss) on macroinvertebrate community variables. I illustrate this by summarising the key findings from the preceding chapters in the context of a conceptual model developed as part of the research. Considering these key findings may improve the recovery of benthic communities as a result of remediation from AMD. These ecological principles could also apply to other remediation projects involving streams degraded by other land-use activities.

Once AMD has been successfully treated, the nature of the regional species pool will be the first factor determining the potential composition of a remediated community (Cornell and Lawton 1992). Identifying potential sources of macroinvertebrates is important because it allows practitioners to: 1) determine the range of species which might constitute a new community; 2) identify potential end points of remediation to help determine when a remediation project has been successful (Downs et al. 2002); and 3) aid practitioners in strategically remediating streams. This final point is particularly important because selecting streams that are connected to source streams or in close proximity to them will assist in rapid colonisation and ensure remediation success (see Chapter Two). In streams associated with the opencast mine on the Stockton Plateau, it was identified that environmental variables (in particular pH and metals) explain a high proportion of the variation within the composition of macroinvertebrate communities. The stream's position within the landscape was the next most important (see Chapter Two). However, once AMD stress is removed, it is likely that the streams geographic position within the landscape will become more important. This is because the cause of the degradation (AMD), which is having the strongest effect, will be removed and natural environmental conditions and processes that influence taxa linkages between streams such as topography and dispersal abilities will then dominate. If all the streams on the Stockton Plateau are assumed to comprise the potential "pool" of re-colonists, then meta-community theory can be used to help predict macroinvertebrate community recovery and community composition. As discussed in Chapter 2, un-impacted streams in the mine landscape have the potential to become donor populations of colonists, whereas treated streams (with chemical conditions within tolerable ranges) will become recipient streams. Consequently, meta-community dynamics will shift from assemblages dominated by a combination of species-sorting and mass effects to assemblages solely influenced by mass effects; thus, dispersal and factors influencing dispersal will become the first level of filters influencing remediation success.

In Chapter 3 I examined longitudinal dispersal, lateral dispersal, the effect of vegetation on lateral dispersal in addition to the relative importance of drift and aerial flight for the supply of macroinvertebrate colonists. Comparisons between upstream and downstream dispersal showed no difference in flight direction. In addition, I found that vegetation type did not significantly influence the densities of adult insects away from the stream. Despite not finding a significant influence of vegetation type, distinct patterns emerged. Scrub vegetation supported higher densities of adult aquatic insects further away from the stream compared to open bedrock where densities declined rapidly and forest where aquatic adult insects appeared to be unable to penetrate the forest in high numbers. When remediating streams the degree of connectivity between unimpacted and AMD treated streams (i.e. if source sites are disconnected or connected to treated sites) is crucial. If un-impacted source population streams are disconnected from remediated reaches it will be necessary for adult aquatic insects to disperse overland to re-colonise remediated reaches. However, my adult dispersal results indicate that if streams are separated by forest or open bedrock the frequency of individuals reaching disconnected streams may be limited compared to scrub. Therefore, these results suggest that scrub vegetation communities should be remediated as fast as practical during mine landscape remediation to facilitate inter-stream dispersal.

In Figure 5.2 I identified that depending on stream connectivity, longitudinal or lateral dispersal will be the most important dispersal pathways for macroinvertebrate recolonisation. If streams are connected to source populations via a headwater or tributary, larvae will be able to drift into remediated reaches while adult insects will be able to fly along the stream corridor to disperse their eggs. Drifting larvae are unlikely to come across any dispersal barriers while aerial flight could potentially be restricted by culverts and bridges, or diverted by anthropogenic light sources if these are present (Blakely and Harding 2005). In contrast, if source populations of macroinvertebrates are disconnected from streams treated for AMD, recolonists will be required to disperse laterally between the streams. However, dispersing insects will have to content with potential dispersal barriers such as topography, vegetation, climatic conditions and individual taxon dispersal abilities. These barriers could potentially filter out species from recolonising streams, thus restricting the available species pool and potentially limit macroinvertebrate recovery. Therefore, while it is context dependant, Figure 5.2 suggests that having a connected stream with source populations of macroinvertebrates will enhance macroinvertebrate recovery as there are fewer filters influencing this dispersal mode.

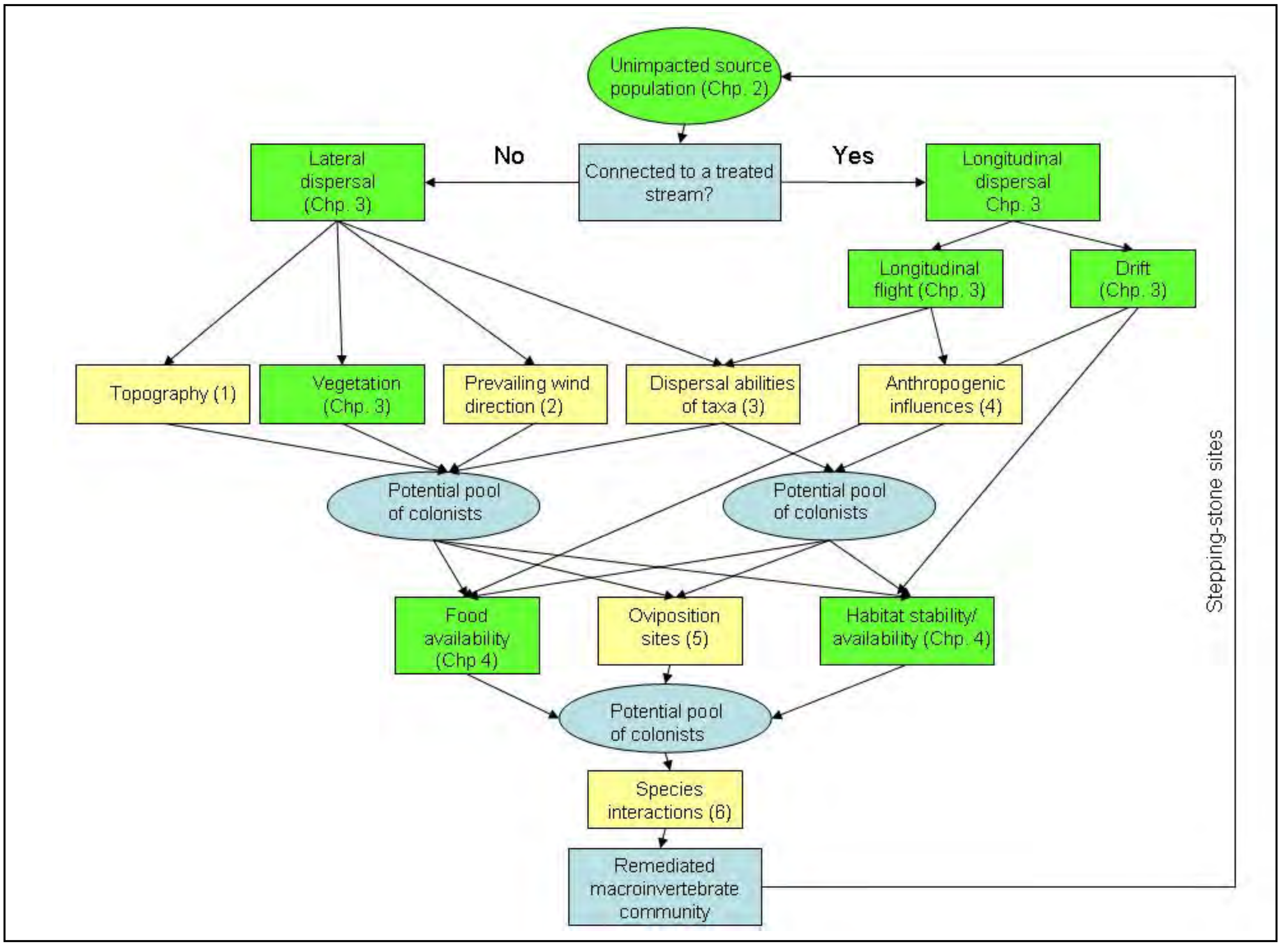

Figure 5.2: Conceptual diagram showing "filters" that affect macroinvertebrate community assembly. These filters should be considered and where possible could be manipulated to assist in the rapid recovery of benthic communities. Green boxes are the factors that I have investigated. Sand coloured boxes are those that I did not consider, but the literature indicates their importance. Supporting references: (1) Wilcock et al. (2003), Finn et al. (2006), Lehrian et al. (2009); (2) Briers et al. (2004), Edde et al. (2006); (3) Winterbourn and Crowe (2001), Briers et al. (2004), Wintebourn (2005); (4) Blakely and Harding 2005; (5) Reich and Downes (2003), Blakely and Harding (2005); (6) McMurtrie (2000), McIntosh (2002), Greig and McIntosh (2006).

Food webs define functional relationships of stream ecosystems (Allan and Castillo 2007), and when they are disrupted through opencast mining, there are often significant changes in the structure and function of the stream community (Schultheis et al. 1997, Carlisle and Clements 2005, Iwasaki et al. 2009). Firstly, the loss of basal food resources such as autochthonous and allochthonous resources can have significant bottom-up effects on macroinvertebrate and ultimately fish communities (Wallace et al. 1997, Hamsher et al. 2004, McIntosh et al. 2005, Entrekin et al. 2007, Bray et al. 2008, Lear et al. 2009). To establish the importance of allochthonous inputs in aiding the remediation of macroinvertebrate communities, I added organic matter to un-impacted streams and measured the macroinvertebrate community response. In Chapter 4, I report trends in the relationship between CPOM and macroinvertebrate community variables over a six month period. However, I was unable to detect a significant increase in benthic CPOM quantity despite adding significantly more organic matter to the streams. The only macroinvertebrate community variables that responded significantly to organic matter additions were total taxa and the percent abundance of collector-browsers. These significant changes however, were linked to a decrease in abundances for both variables between the preliminary survey and the mid-experiment survey. This may have been the result of flooding which exported a number of taxa out of the system as well as CPOM that the organic matter additions may have provided (Quinn et al. 2007, Cordova et al. 2008). Due to the disturbance and geomorphic limitations of Stockton and Denniston Plateau streams, it is possible CPOM is an inappropriate food resource to enhance in these systems. Rather, due to the feeding strategies of the dominant functional feeding group (collector-browsers) it is best to re-introduce biofilm resources and FPOM (Cowie, 1980, Winterbourn et al. 1981, Rounick and Winterbourn 1983). The success of reintroducing the FPOM will likely be dependant on the recovery of bryophytes as these will be instrumental in retaining FPOM due to frequent flooding. Secondly, the recovery of FPOM resources may take a number of years given the requirement for CWD and CPOM to break down to the smaller size fraction.

Additionally, due to the effect of flood disturbance in Stockton Plateau streams, it will take time for sufficient quantities of FPOM to build up in bryophytes to sufficient levels for macroinvertebrates to detect a response and adjust their population size accordingly. Consequently, as shown in Figure 5.2, the availability of food resources acts as an additional filter to the macroinvertebrate recolonists. This is because the lack of food resources will prevent recolonising taxa residing in the remediated stream as they will drift out of the stream in search of more abundant food supplies. Thus, the recolonisation of biofilm and the restoration of FPOM sources (riparian vegetation) should improve the success of macroinvertebrate recovery by providing an important resource. However, while the replanting of riparian vegetation will provide eventual sources of FPOM, the recovery of biofilm will be dependant on the connectivity between remediated streams and unimpacted streams. Biofilm will be required to disperse into remediated reaches to recolonise and they only way that this can occur is if unimpacted streams are connected to recovering streams and the biofilm drifts in. If streams are disconnected, practitioners will have to implement some of the recommendations I have made in Table 5.1.

In some systems restoring riparian vegetation will also improve remediation success via the provision of terrestrially derived insects, which can be consumed by some top predators (Nakano et al. 1999, Konishi et al. 2001, Ruetz Iii et al. 2002, Kawaguchi et al. 2003, Boyero et al. 2008) and may reduce predation pressure on aquatic macroinvertebrates (Nakano et al. 1999). On the other hand, the restoration of aquatic macroinvertebrates can enhance terrestrial predator recovery (e.g. birds and arthropod predators) by providing an additional source of food (Paetzold and Tockner 2005, Fukui et al. 2006, Chan et al. 2007, Iwata 2007, Burdon and Harding 2008).

Another local scale consideration is habitat. Habitat structure is important for macroinvertebrates to avoid predation and disturbance (Suren 1991, Death 2000, Heatherly et al. 2007, Lester et al. 2007, Heino and Korsu 2008, Schneider and Winemiller 2008) and for retaining organic matter (Haapala et al. 2003, Negishi and Richardson 2003, Lepori et al. 2005b). Protruding objects such as emergent boulders, roots or macrophytes act as habitat and are also important for retaining organic matter within streams (Death 2000). Typically, when re-introducing habitat in stream remediation projects only iconic fish species have been considered and usually only adult fish. Now, the life-histories of fish and their differing habitat requirements are being considered. Only recently, have the habitat requirements for macroinvertebrates been considered (Lake et al. 2007). As both the quantity and quality of habitat are degraded during open cast mining I examined the benefits of re-introducing one commonly lost habitat (on the Stockton and Denniston Plateau), bryophytes, in remediated streams. In particular I was interested in how macroinvertebrates would respond to enhanced moss cover.

As expected, the artificial moss was colonised by macroinvertebrates in similar densities as natural bryophyte cover. This suggests that bryophytes are an important habitat in these systems and should be remediated as quickly as possible to assist in the recovery of macroinvertebrate communities. As bryophytes retain organic matter (primarily FPOM) and allow biofilm to colonise they are an important provider of both habitat and food resources that are vital for macroinvertebrates. Also, beneath my artificial tiles to which artificial moss was attached, larger numbers of mayflies and crayfish were observed. This finding suggests that stable habitat and refugia may have been limited for these taxa. Furthermore, significantly enhancing coarse woody debris did increase the number of macroinvertebrates colonising my CWD. However, abundances were still significantly lower than those on the benthic substrate possibly because of the instability of the CWD. Nevertheless, adding wood will provide stable habitat, especially in streams that have experienced severe sedimentation as suggested by Benke et al. (1985) where CWD was responsible for over 80 % of macroinvertebrate production in mud bottom streams despite CWD making up less than 10 % of the available habitat. Other studies, primarily in forestry and agricultural systems have found similar trends although they are not as pronounced as those by Benke et al. (1985) (e.g. Collier and Halliday 2000, Collier and Smith 2003, Lester et al. 2007, Lester and Boulton 2008). Therefore, remediating habitat in streams is important filter that if not considered will prevent recolonisation by appropriate macroinvertebrates (Figure 5.2). This is because habitat provides refuge for macroinvertebrates from flooding (Death and Winterbourn 1994, Milisa et al. 2006), protection from the light (Hughes 1966) and supports the colonisation and retention of important food resources as such biofilm and FPOM (Winterbourn 1982, Suren 1991, Wallace et al. 1999).

Stream remediation plan

To increase the likelihood of benthic community recovery I have compiled a number of recommendations (Table 5.1). These recommendations are generic and could be used in a number of different remediation contexts. Specifically to the streams on the Stockton Plateau I have identified the following recommendations that could be implemented.

1.) It is important that high diversity streams are preserved on the Stockton Plateau to act as sources of colonists for streams remediated from AMD. I highly recommend that the streams MT6, Repo, T69, T35, Stream 2, and M1 are protected and sediment control in Weka Creek is improved. These streams were identified in my large scale spatial survey as being unimpacted (Appendix 1) and are the most important source streams on the Stockton Plateau. MT6, Repo and Weka Creek are all tributaries of the Mangatini Stream. T69 and T35 are tributaries of St Patricks Stream while M1 is in the headwaters of the Mine Creek network, and Stream 2 can be directed into Plover Stream which drains into St Patricks Stream. All these sites are vital because they are all connected to the main streams draining the Stockton Plateau and could allow macroinvertebrates to drift directly in to key AMD remediated waterways. In contrast, if streams are disconnected then re-colonisation will be via aerial re-colonisation of adult aquatic insects (Figure 5.2). This mode is unlikely to be rapid due to the potential effects of complex terrain on the Stockton Plateau, extreme wind conditions and the influence of vegetation.

If a remediated reach is disconnected from sources of re-colonists it is possible to “seed” the stream as I have described in Table 5.1. However, this might have more success if habitat and food resources have been improved so that the “seeded” macroinvertebrates can exploit these resources to survive and reproduce.

2.) One possible option to enhance remediation success is improve habitat and food resources in unimpacted streams so that higher macroinvertebrate populations are supported in these streams. As population sizes increase, drift should also increase, thus providing more drifting re-colonists. For example, M1 is a headwater tributary in the AMD impacted Mine Creek stream network which currently has low densities of desirable EPT taxa. To enhance the densities of EPT taxa moss habitat could be improved, or cobble/boulder substrate added to improve habitat availability. These could also retain organic matter. Nevertheless, due to the flood disturbance, it is important that the habitat is sustainable and appropriately located to withstand the effects of frequent flooding.

3.) In AMD impacted streams on the Stockton Plateau there is a paucity of complex habitat. In my habitat manipulations I was unable to detect a significant positive response of the macroinvertebrate community. However, with an absence of appropriate habitat to provide refugia from flooding macroinvertebrate recovery will be limited in these streams. However, I have not been able to find any literature that has considered re-introducing bryophytes into degraded bedrock streams, but in Table 5.1 I have identified some potential methods. Nevertheless, this is likely to be a slow process and trials should be implemented immediately to identify the most appropriate methods of bryophyte reintroduction. When trialling these methods it would be best to avoid the rainy season to reduce the chance of flooding impacting on the initial re-introduction period.

Table 5.1: This table summarises potential strategies that can be used to enhance the recovery of benthic communities (as well as fish populations) in streams treated for AMD. Such recommendations are also applicable to other stream remediation projects impacted by other land uses.

Remediation considerations	Description of options and/ or importance	Supporting references
• Identify sources of macroinvertebrates and other desirable taxa	An important first step because it enables practitioners to identify where sources of desirable re-colonists are. It also allows practitioners to identify stream community structure and composition, which can suggest potential target endpoints for remediation.	
• Identify appropriate target endpoints for remediation and monitor remediation success	It is vital to identify endpoints for remediation projects. These endpoints are goals that the remediation project aims to achieve. They can be in the form of returning a desired iconic species, or producing a self-sustaining and functional community. It is also important to implement a monitoring program to determine the success or otherwise of the remediation plan. Such a monitoring program can identify issues that are inhibiting recovery of the community and rectifying them. It is also important to be open minded and pragmatic in the approach to implementing new ideas and allow for trial and error so remediation success can be guaranteed.	
• Prioritise stream remediation	Remediation should be focused on streams that are connected to intact/unimpacted headwaters or tributaries. This is to allow desired macroinvertebrates and other freshwater fauna to drift or actively disperse into remediated sections. Strategically selecting these connected streams will enhance remediation success and these	

	remediated streams can subsequently become “stepping stones” to other streams.	
• Actively re-introduce/translocate desired taxa if required	If sites are disconnected from sources of re-colonists and aerial dispersal is considered slow, impossible or undesirable as a re-colonisation mode, an alternative is to “seed” the remediated reach. This could involve placing leaf packs, artificial mosses and/ or cobble filled colonisation baskets into reference streams and leaving them there for sufficient time to allow macroinvertebrate to colonise. Once this has occurred, the colonisation baskets and leaf packs can be placed in the remediate reach and over time these re-colonists should actively disperse into other habitat in the remediated reach.	
• Indentify habitat requirements and remediate	As different taxa have different habitat requirements and preferences it is important to re-introduce all the habitat requirements for the communities that practitioners wish to remediate. These requirements are diverse for different species so this is why it is vital that appropriate target end points are identified.	
• Remediate riparian vegetation and food resources	Re-introducing riparian vegetation will help stabilise stream banks, improve temperature and light penetration regimes through additional shading in streams, and provides sources of CWD, CPOM and terrestrial insects. It provides beneficial habitat and food resources for shredders and ultimately collector-browsers and filter-feeders.	(Nakano et al. 1999, Lepori et al. 2005a, Lepori et al. 2005b)

	Re-introducing periphyton into streams will provide an autotrophic food resource for macroinvertebrates. Periphyton can be sourced from intact headwaters or tributaries. If there are no connected sources "seeding" of periphyton can be conducted. This can be achieved by introducing substrate such as cobble baskets into reference streams and after sufficient time has passed to allow periphyton to establish, remove them and place them in remediated streams.	
• Remove dispersal barriers and improve dispersal facilitators	Culverts have been shown to restrict longitudinal dispersal of macroinvertebrates and prevent fish passage. Removing culverts after remediation is strongly recommended. Where possible, replace them with bridges or larger culverts that are flush with the streambed, or place ramps in front of them to aid fish passage. Adult aquatic insects are attracted to light. To prevent dispersing aquatic adult insects moving towards these lights as opposed to staying in the stream to lay eggs lights should be modified so they are not visible from the stream. Lights could be used to attract adult insects to remediated streams.	(Blakely and Harding 2005, Resh 2005, Hicks et al. 2008)
• Enhance emergence and oviposition sites	Emergence and oviposition sites for certain species can be reduced or eliminated from streams when they are degraded. Re-introducing such sites (e.g. boulders) can improve the re-colonisation and reproductive success of	(Reich and Downes 2003, Blakely and Harding

	insects. Boulders can also provide habitat for macroinvertebrates and act as retentive structures to retain organic matter.	2005)
• Remediate ecosystem processes	Fungal and bacterial processes can be degraded during catchment development, particularly by AMD. As fungal and bacterial processes are important for breaking down organic matter that can be utilised subsequently by macroinvertebrates belonging to other functional feeding groups it would be desirable to re-introduce fungi and bacteria. This could potentially be achieved by collected decaying organic matter from reference streams and putting this in retentive areas within remediated streams. However, this is unlikely to work in flood prone streams as this material will be frequently flushed out.	
• Control episodic pollution and consider disturbance events	It is important that any pollutants that caused the original degradation are prevented from entering streams again (even episodically) as they can degrade stream ecosystems again and eliminate all the progress that may have been made in the recovery phase. It is also important to consider the effect of disturbance events, especially in the context of climate change. Floods and drought can limit recovery and periodically "re-set" any advances that have been made. Additionally, the frequency and severity of extreme events may increase with climate change thus providing a limit to recovery.	(MacCausland and McTammany 2007)

- Species loss, introduced species and biotic interactions	Due to the initial degradation, species may have been lost and biotic interactions may have changed. The introduction of other non-native species may also change biotic interactions. Therefore, community structure and composition may not necessarily be the same as found in the pre-disturbance state. Depending on the situation there may be some solutions. For example, if an introduced fish has invaded it may be possible to remove it This highlights the importance of identifying endpoints for remediation to determine when remediation has been successful because introduced species or species extinction prevent the pre-disturbance community being restored. Thus, remediation should aim at remediating a functional community that's structure may not be the same.	(Lake et al. 2007)

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References

Alarcon, L. E. 1997. Long term mine site rehabilitation studies at Stockton open-cast coal mine. MSc. University of Canterbury, Christchurch.

Aldridge, K. T., J. D. Brookes, and G. G. Ganf. 2009. Rehabilitation of stream ecosystem functions through the reintroduction of coarse particulate organic matter. Restoration Ecology **17**:97-106.

Allan, J. D. and M. M. Castillo. 2007. Stream ecology: structure and function of running waters. 2nd Edition. Springer, Dordrecht.

Allard, M. and G. Moreau. 1987. Effects of experimental acidification on a lotic macroinvertebrate community. Hydrobiologia **144**:37-49.

Anholt, B. R. 1995. Density-dependence resolves the stream drift paradox. Ecology **76**:2235-2239.

Anthony, M. K. 1999. Ecology of streams contaminated by acid mine drainage, near Reefton, South Island. MSc. University of Canterbury, Christchurch.

APHA. 2005. Standard methods for the examination of water and waste water. 21st edition. American Public Health Association, Washington.

Armitage, P. D., M. J. Bowes, and H. M. Vincent. 2007. Long-term changes in macroinvertebrate communities of a heavy metal polluted stream: the river Nent (Cumbria, UK) after 28 years. River Research and Applications **23**:997-1015.

Armstrong, P. 1996. The effects of streambed stability and flow variability on the production of aquatic invertebrates. MSc. University of Canterbury, Christchurch.

Baer, S. G., E. R. Siler, S. L. Eggert, and J. B. Wallace. 2001. Colonization and production of macroinvertebrates on artificial substrata: upstream-downstream responses to a leaf litter exclusion manipulation. Freshwater Biology **46**:347-365.

Barnden, A. 2005. Ecology of streams affected by iron precipitates and iron flocculants. MSc. University of Canterbury, Christchurch.

Barnden, A. R. and J. S. Harding. 2005. Shredders and leaf breakdown in streams polluted by coal mining in the South Island, New Zealand. New Zealand Natural Sciences **30**:35-48.

Barry, K. L., J. A. Grout, C. D. Levings, B. H. Nidle, and G. E. Piercey. 2000. Impacts of acid mine drainage on juvenile salmonids in an estuary near Britannia Beach in

Howe Sound, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences **57**:2032-2043.

Battaglia, M., G. C. Hose, E. Turak, and B. Warden. 2005. Depauperate macroinvertebrates in a mine affected stream: clean water may be the key to the recovery. Environmental Pollution **138**:132-141.

Bell, H. L. 1971. Effect of low pH on the survival and emergence of aquatic insects. Water Research **5**:313-319.

Beltman, D. J., W. H. Clements, J. Lipton, and D. Cacela. 1999. Benthic invertebrate metals exposure, accumulation and community level effects downstream from a hardrock mine site. Environmental Toxicology and Chemistry **18**:299-307.

Benke, A. C., R. L. Henry, D. M. Gillespie, and R. J. Hunter. 1985. The importance of snag habitat for animal production is southeastern streams. Fisheries **10**:8-13.

Berezina, N. A. 2001. Influence of ambient pH on freshwater invertebrates under experimental conditions Russian Journal of Ecology **32**:343-351.

Berger, W. H. and F. L. Parker. 1970. Diversity of planktonic Foraminifera in deep sea sediments. Science **168**:1345-1347.

Biggs, B. J. F. and C. Kilroy. 2004. Periphyton. Pages 15.01-15.22 *in* J. S. Harding, P. Mosley, C. Pearson, and B. Sorrell, editors. Freshwaters of New Zealand. New Zealand Limnological Society, Christchurch.

Bilton, D. T., J. R. Freeland, and B. Okamura. 2001. Dispersal in freshwater invertebrates: mechanisms and consequences. Annual Review of Ecology and Systematics **3**:159-181.

Bird, G. A. and H. B. N. Hynes. 1981. Movements of adult aquatic insects near streams in Southern Ontario. Hydrobiologia **77**:65-69.

Bis, B. and L. W. G. Higler. 2001. Riparian vegetation of streams and the macroinvertebrate community structure. Ecohydrology and Hydrobiology **1**:253-260.

Blakely, T. J. and J. S. Harding. 2005. Longitudinal patterns in benthic communities in an urban stream under restoration. New Zealand Journal of Marine and Freshwater Research **39**:17-28.

Bo, T. M., S. Fenoglio, and G. Malacarne. 2006. Colonisation patterns and vertical movements of stream invertebrates in the interstitial zone: a case study in the Apennines, NW Italy. Hydrobiologia **568**:67-78.

Bond, N. R. and B. J. Downes. 2003. The independent and interactive effects of fine sediment and flow on benthic invertebrate communities characteristic of small upland streams. Freshwater Biology **48**:455-465.

Bond, N. R., S. Sabater, A. Glaister, S. Roberts, and K. Vanderkruk. 2006. Colonisation of introduced timber by algae and invertebrates, and its potential role in aquatic ecosystem restoration. Hydrobiologia **556**:303-316.

Bonnett, M. L. and J. R. E. Sykes. 2002. Habitat preferences of giant kokopu, *Galaxias argenteus*. New Zealand Journal of Marine and Freshwater Research **36**:13-24.

Boyero, L., P. A. Rincoǹn, and R. G. Pearson. 2008. Effects of a predatory fish on a tropical detritus-based food web. Ecological Research **23**:649-655.

Bray, J. P., P. A. Broady, D. K. Niyogi, and J. S. Harding. 2008. Periphyton communities in New Zealand streams impacted by acid mine drainage. Marine and Freshwater Research **59**:1084-1091.

Briers, R. A., H. M. Cariss, and J. H. R. Gee. 2003. Flight activity of adult stoneflies in relation to weather. Ecological Entomology **28**:31-40.

Briers, R. A., J. H. R. Gee, H. M. Cariss, and R. Geoghegan. 2004. Inter-population dispersal by adult stoneflies detected by stable isotope enrichment. Freshwater Biology **49**:425-431.

Brinkman, S. F. and W. D. Johnston. 2008. Acute toxicity of aqueous copper, cadmium, and zinc to the mayfly *Rhithrogena hageni*. Archives of Environmental Contamination and Toxicology **54**:466-472.

Brusven, M. A., R. M. William, and R. C. Biggam. 1990. The role of aquatic moss on community composition and drift of fish-food organisms. Hydrobiologia **196**:39-50.

Bunn, S. E. and J. M. Hughes. 1997. Dispersal and recruitment in streams: Evidence from genetic studies. Journal of the North American Benthological Society **16**:338-346.

Burdon, F. J. and J. S. Harding. 2008. The linkage between riparian predators and aquatic insects across a stream-resource spectrum. Freshwater Biology **53**:330-346.

Caceres, C. E. and D. A. Soluk. 2002. Blowing in the wind: field test of overland dispersal and colonisation by aquatic invertebrates. Oecologia **131**:402-408.

Caquet, T., M. L. Hanson, M. Roucaute, D. W. Graham, and L. Lagadic. 2007. Influence of isolation on the recovery of pond mesocosms from the application of an insecticide. II. Benthic macroinvertebrate responses. Environmental Toxicology and Chemistry **26**:1280-1290.

Carlisle, D. M. and W. H. Clements. 2005. Leaf litter breakdown, microbial respiration and shredder production in metal-polluted streams. Freshwater Biology **50**:380-390.

Chadwick, J. W., S. P. Canton, and R. L. Dent. 1986. Recovery of benthic invertebrate communities in Silver Bow Creek, Montana, following improved metal mine wastewater treatment. Water, Air, and Soil Pollution **28**:427-438.

Chan, E. K. W., Y. Zhang, and D. Dudgeon. 2007. Contribution of adult aquatic insects to riparian prey availability along tropical forest streams. Marine and Freshwater Research **58**:725-732.

Chaput-Bardy, A., C. Lemaire, D. Picard, and J. Secondi. 2008. In-stream and overland dispersal across a river network influences gene flow in a freshwater insect, *Calopteryx splendens*. Molecular Ecology **17**:3496-3505.

Cherry, D. S., R. J. Currie, D. J. Soucek, H. A. Latimer, and G. C. Trent. 2001. An interactive assessment of a watershed impacted by abandoned mined land discharges. Environmental Pollution **111**:377-388.

Clark, J. L. and W. H. Clements. 2006. The use of in situ and stream mesocosm experiments to assess population- and community-level responses to metals. Environmental Toxicology and Chemistry **25**:2306-2312.

Clements, W. H. 1994. Benthic invertebrate community responses to heavy metals in the Upper Arkansas River Basin, Colorado. Journal of the North American Benthological Society **13**:30-44.

Clements, W. H. 1999. Metal tolerance and predator-prey interactions in benthic macroinvertebrate stream communities. Ecological Applications **9**:1073-1084.

Cole, M. B., D. E. Arnold, and B. J. Watten. 2001. Physiological and behavioural responses of stonefly nymphs to enhanced limestone treatment of acid mine drainage Water Research **35**:625-632.

Collier, K. J., O. J. Ball, A. K. Graesser, M. R. Main, and M. J. Winterbourn. 1990. Do organic and anthropogenic acidity have similar effects on aquatic fauna? Oikos **59**:33-38.

Collier, K. J. and J. N. Halliday. 2000. Macroinvertebrate-wood associations during decay of plantation pine in New Zealand pumice-bed streams: stable habitat or trophic subsidy? Journal of the North American Benthological Society **19**:94-111.

Collier, K. J. and J. M. Quinn. 2003. Land-use influences macroinvertebrate community response following a pulse disturbance. Freshwater Biology **48**:1462-1481.

Collier, K. J. and B. J. Smith. 1995. Sticky trapping of adult mayflies, stoneflies and caddisflies alongside three contrasting streams near Hamilton, New Zealand. New Zealand Natural Sciences **22**:1-9.

Collier, K. J. and B. J. Smith. 1997. Dispersal of adult caddisflies (Trichoptera) into forests alongside three New Zealand streams. Hydrobiologia **361**:53-65.

Collier, K. J. and B. J. Smith. 1998. Dispersal of adult caddisflies (Trichoptera) into forests alongside three New Zealand streams. Hydrobiologia **361**:53-65.

Collier, K. J. and B. J. Smith. 2003. Role of wood in pumice-bed streams II: Breakdown and colonisation. Forest Ecology and Management **177**:261-276.

Collier, K. J., B. J. Smith, and N. J. Halliday. 2004. Colonization and use of pine wood versus native wood in New Zealand plantation forest streams: Implications for riparian management. Aquatic conservation: Marine and Freshwater ecosystems **14**:179-199.

Collier, K. J. and M. J. Winterbourn. 1987. Faunal and chemical dynamics of some acid and alkaline New Zealand streams. Freshwater Biology **18**:227-240.

Collier, K. J. and M. J. Winterbourn. 1990. Population dynamics and feeding of mayfly larvae in some acid and alkaline New Zealand streams. Freshwater Biology **23**:181-189.

Compson, Z. G., M. Z. Mier, and J. C. Marks. 2009. Effects of travertine and flow on leaf retention in Fossil Creek, Arizona. Hydrobiologia **630**:187-197.

Cooling, M. P. and A. J. Boulton. 1993. Aspects of the hyporheic zone below the terminus of a South Australian arid-zone stream. Australian Journal of Marine and Freshwater Research **44**:411-426.

Cordova, J. M., E. J. Rosi-Marshall, J. L. Tank, and G. A. Lamberti. 2008. Coarse particulate organic matter transport in low-gradient streams of the Upper Peninsula of Michigan. Journal of the North American Benthological Society **27**:760-771.

Cornell, H. V. and J. H. Lawton. 1992. Species interactions, local and regional processes, and limits to the richness of ecological communities-a theoretical perspective. Journal of Animal Ecology **61**:1-12.

Cottam, D. P. 1999. The importance of riparian cover and allochthonous inputs for the biological health of Christchurch waterways. MSc. University of Canterbury, Christchurch.

Cottenie, K. 2005. Integrating environmental and spatial processes in ecological community dynamics. Ecology Letters **8**:1175-1182.

Courtney, L. A. and W. H. Clements. 1998. Effects of acidic pH on benthic macroinvertebrate communities in stream microcosms. Hydrobiologia **379**:135-145.

Courtney, L. A. and W. H. Clements. 2000. Sensitivity to acidic pH in benthic invertebrate assemblages with different histories to exposure to metals. Journal of the North American Benthological Society **19**:112-127.

Courtney, L. A. and W. H. Clements. 2002. Assessing the influence of water and substratum quality on benthic macroinvertebrate communities in a metal-polluted stream: an experimental approach. Freshwater Biology **47**:1766-1778.

Cover, M. R., C. L. May, W. E. Dietrich, and V. H. Resh. 2008. Quantitative linkages among sediment supply, streambed fine sediment and benthic macroinvertebrates in Northern California streams. Journal of the North American Benthological Society **27**:135-149.

Cowie, B. 1980. Community dynamics of the benthic fauna in a West Coast stream ecosystem. PhD. University of Canterbury, Christchurch.

Cowie, B. 1983. Macroinvertebrate taxa from a southern New Zealand montane stream continuum. New Zealand Entomologist **7**:439-447.

Dangles, O., B. Malmqvist, and H. Lauden. 2004. Naturally acidic freshwater ecosystems are diverse and functional: evidence from boreal streams. Oikos **104**:149-155.

de Castro Vasconcelos, M. and A. Melo. 2008. An experimental test of the effects of inorganic sediment addition on benthic macroinvertebrates of a subtropical stream. Hydrobiologia **610**:321-329.

De Jonge, M., B. Van de Vijver, R. Blust, and L. Bervoets. 2008. Responses of aquatic organisms to metal pollution in a lowland river in Flanders: a comparison of diatoms and macroinvertebrates. Science of the Total Environment **407**:615-629.

de la Pena, S. and R. Barreiro. 2009. Biomonitoring acidic drainage impact in a complex setting using periphyton. Environmental Monitoring and Assessment **150**:351-363.

Death, R. G. 2000. Invertebrate-substrate relationships. Pages 157-178 *in* K. J. Collier and M. J. Winterbourn, editors. New Zealand Stream Invertebrates: ecology and Implications for Management. New Zealand Limnological Society, Christchurch.

Death, R. G. 2004. Patterns of spatial resource use in lotic invertebrate assemblages. Hydrobiologia **513**:171-182.

Death, R. G. and M. J. Winterbourn. 1994. The measurement of environmental stability in streams: a multivariate approach. Journal of the North American Benthological Society **13**:125-139.

Death, R. G. and E. M. Zimmermann. 2005. Interaction between disturbance and primary productivity in determining stream invertebrate diversity. Oikos **111**:392-402.

Delettre, Y. R. and N. Morvan. 2000. Dispersal of adult aquatic Chironomidae (Diptera) in agricultural landscapes. Freshwater Biology **44**:399-411.

DeNicola, D. M. and M. G. Stapleton. 2002. Impact of acid mine drainage on benthic communities in streams: the relative roles of substratum vs. aqueous effects. Environmental Pollution **119**:303-315.

Dewsen, Z. S., A. B. W. James, and R. G. Death. 2007. A review of the consequences of decreased flow for instream habitat and macroinvertebrates Journal of the North American Benthological Society **26**:401-415.

Doi, H. 2009. Spatial patterns of autochthonous and allochthonous resources in aquatic food webs. Population Ecology **51**:57-64.

Doi, H., A. Takagi, and E. Kikuchi. 2007. Stream macroinvertebrate community affected by point-source metal pollution. International Review of Hydrobiology **92**:258-266.

Downs, P. W. and G. M. Kondolf. 2002. Post-project appraisals in adaptive management of river channel restoration. Environmental Management **29**:477-496.

Downes, B. J., P. S. Lake, S. G. Schreiber, and A. Glaister. 2000. Habitat structure, resources and diversity: the separate effects of surface roughness and macroalgae on stream invertebrates. Oecologia **123**:569-581.

Dsa, J. V., K. S. Johnson, D. Lopez, C. Kanuckel, and J. Tumlinson. 2008. Residual toxicity of acid mine drainage- contaminated sediment to stream macroinvertebrates: relative contribution of acidity vs metals. Water, Air, Soil Pollution **194**:185-197.

Dudley, T. and N. H. Anderson. 1982. A survey of invertebrates associated with wood debris in aquatic habitats. Melanderia **39**:1-21.

Duran, M. 2006. Field experiment on drift and colonization of benthic macroinvertebrates in Gokptnar Stream (Denizli, E Turkey). Pakistan Journal of Biological Sciences **9**:493-496.

Edde, P. A., T. W. Phillips, C. Nansen, and M. E. Payton. 2006. Flight activity of the lesser grain borer, *Rhyzopertha dominic*a F. (Coleoptera : Bostrichidae), in relation to weather. Environmental Entomology **35**:616-624.

Eggert, S. L. and J. B. Wallace. 2003a. Litter breakdown and invertebrate detritivores in a resource-depleted Appalachian stream. Archiv für Hydrobiologie **156**:315-338.

Eggert, S. L. and J. B. Wallace. 2003b. Reduced detrital resources limit *Pycnopsyche gentitis* (Trichoptera: Limnephilidae) production and growth. Journal of the North American Benthological Society **22**:388-400.

Ehrman, J. M., F. Balrlocher, R. Wennrich, G. J. Krauss, and G. Krauss. 2008. Fungi in a heavy metal precipitating stream in the Mansfeld mining district, Germany. Science of the Total Environment **389**:486-496.

Ehrman, T. P. and G. A. Lamberti. 1992. Hydraulic and particulate matter retention in a 3rd-order Indiana stream. Journal of the North American Benthological Society **11**:341-349.

Elenter, D., K. Milferstedt, W. Zhang, M. Hausner, and E. Morgenroth. 2007. Influence of detachment on substrate removal and microbial ecology in a heterotrophic/autotrophic biofilm. Water Research **41**:4657-4671.

Elliott, J. M. 2003. A comparative study of the dispersal of 10 species of stream invertebrates. Freshwater Biology **48**:1652-1668.

Entrekin, S. A., J. L. Tank, E. J. Rosi-Marshall, T. J. Hoellein, and G. A. Lamberti. 2008. Responses in organic matter accumulation and processing to an experimental wood addition in 3 headwater streams. Freshwater Biology **53**:1642-1657.

Entrekin, S. A., J. B. Wallace, and S. L. Eggert. 2007. The response of Chironomidae (Diptera) to a long-term exclusion of terrestrial organic matter. Hydrobiologia **575**:401-413.

Erskine, W. D. and A. A. Webb. 2003. Desnagging to resnagging: new directions in river rehabilitation in south eastern Australia. River Research and Applications **19**:233-249.

Everett, R. A. and G. M. Ruiz. 1993. Coarse woody debris as a refuge from predation in aquatic communities - an experimental test. Oecologia **93**:475-486.

Feldman, R. S. and E. F. Connor. 1992. The relationship between pH and community structure of invertebrates in streams of the Shenandoah National Park, Virginia, U.S.A. Freshwater Biology **27**:261-276.

Fenoglio, S., P. Agosta, T. Bo, and M. Cucco. 2002. Field experiments on colonisation and movements of stream invertebrates in an Apennine river (Visone, NW Italy). Hydrobiologia **474**:125-130.

Finn, D. S., M. S. Blouin, and D. A. Lytle. 2007. Population genetic structure reveals terrestrial affinities for a headwater stream insect. Freshwater Biology **52**:1881-1897.

Finn, D. S., D. M. Theobald, W. C. Black, and N. L. Poff. 2006. Spatial population genetic structure and limited dispersal in a rocky mountain alpine stream insect. Molecular Ecology **15**:3553-3566.

Fowler, R. T. 2002. Relative importance of surface and subsurface movement on benthic community recovery in the Malaretu River, North Island, New Zealand. New Zealand Journal of Marine and Freshwater Research **36**:459-469.

Fraser, I. A. 2006. The effects of forest fragmentation on stream invertebrate communities on Banks Peninsula. MSc. University of Canterbury, Christchurch.

Frick, K. G. and J. Herrmann. 1990aAluminium and pH effects on sodium ion regulation in mayflies. Pages 409-412 *in* B. J. Mason, editor. The surface waters acidification program. Cambridge University Press, London.

Frick, K. G. and J. Herrmann. 1990b. Aluminium accumulation in a lotic mayfly at low pH-A laboratory study. Ecotoxicology and Environmental Safety **19**:81-88.

Fukui, D., M. Murakami, S. Nakano, and T. Aoi. 2006. Effect of emergent aquatic insects on bat foraging in a riparian forest. Journal of Animal Ecology **75**:1252-1258.

Gandy, C. J., P. L. Younger, and P. Lindsay. 2009. Modelling waste rock leaching at an opencast coal mine in New Zealand: the effects of mineral grain size on long-term contamination. Journal of Environmental Geology **In Press**.

Gayraud, S., M. Philippe, and L. Maridet. 2000. The response of benthic macroinvertebrates to artificial disturbance: drift or vertical movement in the gravel bed of two Sub-Alpine streams? Archiv für Hydrobiologie **147**:431-446.

Gerhardt, A. 1990. Effects of subacute doses of Cd on pH-stressed *Leptophlebia marginata* (L.) and *Baetis rhodani* Pictet (Insecta: Ephemeroptera). Environmental Pollution **67**:29-42.

Gerhardt, A. 1992. Effects of subacute doses of iron (Fe) on *Leptophlebia marginata* (Ephemeroptera: Insecta) Freshwater Biology **27**:79-84.

Gerhardt, A., L. Janssens De Bisthoven, and A. M. V. M. Soares. 2005. Effects of acid mine drainage and acidity on the activity of *Choroterpes picteti* (Ephemeroptera: Leptophlebiidae). Archives of Environmental Contamination and Toxicology **48**:450-458.

Geurold, F., J. P. Boudot, G. Jacquemin, D. Vein, D. Merlet, and J. Rouiler. 2000. Macroinvertebrate community loss as a result of headwater stream acidification in the Vosges Mountains (NE France). Biodiversity Conservation **9**:767-783.

Gibbins, C., D. Vericat, and R. J. Batalla. 2007. When is stream invertebrate drift catastrophic? The role of hydraulics and sediment transport in initiating drift during flood events. Freshwater Biology **52**:2369-2384.

Goovaerts, P. 1997. Geostatistics for natural resources evaluation. Oxford University Press, New York, U.S.A.

Graesser, A. K. 1988. Physico-chemical conditions and benthic community dynamics in 4 South Westland streams. PhD. University of Canterbury, Christchurch, New Zealand.

Gray, N. F. 1998. Acid mine drainage composition and the implications for its impact on lotic systems. Water Resources **32**:2122-2134.

Greene, B. T., W. H. Lowe, and G. E. Likens. 2008. Forest succession and prey availability influence the strength and scale of terrestrial-aquatic linkages in a headwater salamander system. Freshwater Biology **53**:2234-2243.

Greig, H. S. and A. R. McIntosh. 2006. Indirect effects of predatory trout on organic matter processing in detritus-based stream food webs. Oikos **112**:31-40.

Griffith, M. B., E. W. Barrows, and S. A. Perry. 1998. Lateral dispersal of adult aquatic insects (Plecoptera, Trichoptera) following emergence from headwater streams in forested Appalachian catchments. Annals of the Entomological Society of America **91**:195-201.

Haapala, A., T. Muotka, and P. Laasonen. 2003. Distribution of benthic macroinvertebrates and leaf litter in relation to streambed retentivity: implications for headwater stream restoration. Boreal Environment Research **8**:19-30.

Hall, R. J., G. E. Likens, S. B. Fiance, and G. R. Hendrey. 1980. Experimental acidification of a stream in the Hubbard Brook experimental forest, New Hampshire. Ecology **61**:976-989.

Hall, R. O., J. B. Wallace, and S. L. Eggert. 2000. Organic matter flow in stream food webs with reduced detrital resource base. Ecology **81**:3445-3463.

Harbrow, M. A. 2001. Ecology of streams affected by acid mine drainage near Westport, South Island, New Zealand. MSc. University of Canterbury, Christchurch.

Hamsher, S. E., R. G. Verb, and M. L. Vis. 2004. Analysis of acid mine drainage impacted streams using a periphyton index. Journal of Freshwater Ecology **19**:313-324.

Harding, J. and I. Boothroyd. 2004. Impacts of mining. Pages 36.01-36.10 *in* J. Harding, P. Mosley, C. Pearson, and B. Sorrell, editors. Freshwaters of New Zealand. New Zealand Hydrological Society Inc. and New Zealand Limnological Society Inc., Christchurch.

Harding, J. S. 2005. Impacts of metals and mining on stream communities. Pages 343-357 *in* T. A. Moore, A. Black, J. A. Centeno, J. S. Harding, and D. A. Trumm, editors. Metal contaminants in New Zealand. Resoultionz Press, Christchurch.

Harding, J. S. and M. J. Winterbourn. 1995. Effects of contrasting land use on physico-chemical conditions and benthic assemblages of streams in a Canterbury (South Island, New Zealand) river system. New Zealand Journal of Marine and Freshwater Research **39**:479-492.

Hart, R. C. 1985. Seasonality or aquatic invertebrates in low-latitude and southern hemisphere inland waters. Hydrobiologia **125**:151-178.

Hartmann, K. J., M. D. Kaller, H. J.W., and J. A. Sweka. 2005. How much do valley fills influence headwater streams. Hydrobiologia **532**:91-102.

Heatherly, T., M. R. Whiles, T. V. Royer, and M. B. David. 2007. Relationships between water quality, habitat quality on macroinvertebrate assemblages in Illinois streams. Journal of Environmental Quality **36**:1653-1660.

Heckman, C. H. and N. Friberg. 2005. Macroinvertebrate community response to pulse exposure with the insecticide lambda-cyhalothrin using in-stream mesocoms. Environmental Toxicology and Chemistry **24**:582-590.

Heino, J. and K. Korsu. 2008. Testing species-stone area and species bryophyte cover relationships in riverine macroinvertebrates at small scales. Freshwater Biology **53**:558-568.

Heino, J. and H. Mykrä. 2008. Control of stream insect assemblages: roles of spatial configuration and local environmental factors. Ecological Entomology **33**:614-622.

Henley, S. N. 2007. Stockton coal mining licence: mining history- a brief overview. Pages 1-7. Solid Energy New Zealand Inc., Westport, New Zealand.

Herrmann, J. and K. G. Andersson. 1986. Aluminium impacts on respiration of lotic mayflies at low pH. Water, Air, and Soil Pollution **30**:703-709.

Herrmann, J., E. Degerman, A. Gerhardt, C. Johansson, P. E. Lingdell, and I. P. Muniz. 1993. Acid-stress effects on stream biology. Ambio **22**:298-307.

Hershey, A. E., J. Pastor, B. J. Peterson, and G. W. Kling. 1993. Stable isotopes resolve the drift paradox for Baetis mayflies in an Arctic River. Ecology **74**:2315-2325.

Hickey, C. W. and L. A. Golding. 2002. Response of macroinvertebrates to copper and zinc in a stream mesocosm. Environmental Toxicology and Chemistry **21**:1854-1863.

Hicks, B., M. Roper, T. Ito, and D. Campbell. 2008. Passage of inanga (*Galaxias maculatus*) over artificial ramps as a means to restoring upstream access to stream habitat. Waikato University, Hamilton.

Horner, A. J. and C. R. Goldman. 1994. Limnology. McGraw-Hill Inc, Singapore.

Huang, Y., Y. Yan, and X. Li. 2008. Food web structure of benthic macroinvertebrates in a second order stream of the Hanjiang River Basin in middle China. Journal of Freshwater Ecology **23**:421-427.

Hughes, D. A. 1966. The role of responses to light in the selection and maintenance of microhabitat by the nymphs of two species of mayfly. Annual Behaviour **14**:17-33.

Hughes, J. M., S. E. Bunn, D. A. Hurwood, and C. Clearly. 1998. Dispersal and recruitment of *Tasiagma ciliata* (Trichoptera: Tasimiidae) in rainforest streams, south-eastern Australia. Freshwater Biology **39**:117-127.

Hughes, J. M., P. B. Mather, A. L. Sheldon, and F. W. Allendorf. 1999. Genetic structure of the stonefly, *Yoraperla brevis*, populations: the extent of gene flow among adjacent montane streams. Freshwater Biology **41**:63-72.

Hutchens, J. J., J. B. Wallace, and E. D. Romaniszyn. 2004. Role of *Podostemum ceratophyllum* Michx. in structuring benthic macroinvertebrate assemblages in a southern Appalachian river. Journal of the North American Benthological Society **23**:713-727.

Ilmonen, J., L. Paasivirta, R. Virtanen, and T. Muotka. 2009. Regional and local drivers of macroinvertebrate assemblages in boreal springs. Journal of Biogeography **36**:822-834.

Iwasaki, Y., T. Kagaya, K. I. Miyamoto, and H. Matsuda. 2009. Effects of heavy metals on riverine benthic macroinvertebrate assemblages with reference to potential food availability for drift-feeding fishes. Environmental Toxicology and Chemistry **28**:354-363.

Iwata, T. 2007. Linking stream habitats and spider distribution: Spatial variations in trophic transfer across a forest-stream boundary. Ecological Research **22**:619-628.

Jackson, J. K. and V. H. Resh. 1989. Distribution and abundance of adult aquatic insects in forest adjacent to a northern California stream. Environmental Entomology **18**:278-283.

Jansen, M., M. Judas, and J. Saborowski. 2002. Spatial modelling in forest ecology and management: A case study. Springer, Berlin, Germany.

Johnson, L. B., D. H. Breneman, and C. Richards. 2003. Macroinvertebrate community structure and function associated with large wood in low gradient streams. River Research and Applications **19**:199-218.

Kawaguchi, Y., Y. Taniguchi, and S. Nakano. 2003. Terrestrial invertebrate inputs determine the local abundance of stream fishes in a forested stream. Ecology **84**:701-708.

Kelly, M. 1988. Mining and the freshwater environment. Elsevier Science Publishers Ltd., London.

Kiffney, P. M. and W. H. Clements. 1994. Effects of heavy metals on a macroinvertebrate assemblage from a Rocky Mountain stream in experimental microcosms. Journal of the North American Benthological Society **13**:511-523.

Konishi, M., S. Nakano, and T. Iwata. 2001. Trophic cascading effects of predatory fish on leaf litter processing in a Japanese stream. Ecological Research **16**:415-422.

Korsu, K. 2004. Response of benthic invertebrates to disturbance from stream restoration: the importance of bryophytes. Hydrobiologia **523**:37-45.

Kovats, Z., J. Bciborowski, and L. Corkum. 1996. Inland dispersal of adult aquatic insects. Freshwater Biology **36**:265-276.

Kramer, J. F. 1982. Substrate selection, colonisation rates and movements of the benthic invertebrate fauna inhabiting the south branch of the Waimakariri River, New Zealand. MSc. University of Canterbury, Christchurch.

Lake, P. S., N. Bond, and P. Reich. 2007. Linking ecological theory with stream restoration. Freshwater Biology **52**:597-615.

Langford, T. E. L., P. J. Shaw, A. J. D. Frerguson, and S. R. Howard. 2009. Long-term recovery of macroinvertebrate biota in grossly polluted streams: re-colonisation as a constraint to ecological quality. Ecological Indicators **In Press**.

Larsen, S., I. P. Vaughan, and S. J. Ormerod. 2009. Scale-dependant effects of fine sediments on temperate headwater invertebrates Freshwater Biology **54**:203-219.

Ledger, M. E. and A. G. Hildrew. 2005. The ecology of acidification and recovery: changes in herbivore-algal food web linkages across a stream pH-gradient. Environmental Pollution **137**:103-118.

Lehrian, S., S. U. Pauls, and P. Haase. 2009. Contrasting patterns of population structure in the montane caddisflies *Hydropsyche tenuis* and *Drusus discolor* in the Central European highlands. Freshwater Biology **54**:283-295.

Leibold, M. A., M. Holyoak, N. Mouquet, P. Amarasekare, J. M. Chase, M. F. Hoopes, R. D. Holt, J. B. Shurin, R. Law, D. Tilman, M. Loreau, and A. Gonzalez. 2004. The metacommunity concept: a framework for multi-scale community ecology. Ecology Letters **7**:601-613.

Lemly, A. D. and R. H. Hilderbrand. 2000. Influence of large woody debris on stream insect communities and benthic detritus. Hydrobiologia **421**:179-185.

Lepori, F., D. Palm, E. Brännäs, and B. Malmqvist. 2005a. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity. Ecological Applications **15**:2060-2071.

Lepori, F., D. Palm, and B. Malmqvist. 2005b. Effects of stream restoration on ecosystem functioning: detritus retentiveness and decomposition. Journal of Applied Ecology **42**:228-238.

Leps, J. and P. Smilauer. 2003. Multivariate analysis of ecological data using CANOCO. Cambridge University Press, Cambridge, U.S.A.

Leroux, S. J. and M. Loreau. 2008. Subsidy hypothesis and strength of trophic cascades across ecosystems. Ecology Letters **11**:1147-1156.

Lester, R. E. and A. J. Boulton. 2008. Rehabilitating agricultural streams in Australia with wood: a review. Environmental Management **42**:310-326.

Lester, R. E., W. Wright, and M. Jones-Lennon. 2007. Does adding wood to agricultural streams enhance biodiversity? An experimental approach. Marine and Freshwater Research **58**:687-698.

Li, A. O. Y. and D. Dudgeon. 2008. Food resources of shredders and other benthic macroinvertebrates in relation to shading conditions in tropical Hong Kong streams. Freshwater Biology **53**:2011-2025.

Lin, C., X. Tang, W. Lu, L. Yan, Y. Wu, C. Nie, C. Chu, and J. Long. 2005. Environmental impacts of surface mining on mined lands, affected streams and agricultural lands in the Dabaoshan mine region, Southern China. Land Degradation and Development **16**:463-474.

Lin, C., Y. Wu, W. Lu, A. Chen, and Y. Liu. 2007. Water chemistry and ecotoxicity of an acid mine drainage-affected stream in subtropical China during a major flood event. Journal of Hazardous Materials **142**:199-207.

Linklater, W. and M. J. Winterbourn. 1993. Life histories and production of two trichopteran shredders in New Zealand streams with different riparian vegetation. New Zealand Journal of Marine and Freshwater Research **27**:61-70.

Lowe, W., H,. 2002. Landscape-scale spatial population dynamics in human impacted stream systems. Environmental Management **30**:225-233.

MacCausland, A. and M. E. McTammany. 2007. The impact of episodic coal mine drainage pollution on benthic macroinvertebrates in streams in the Anthracite region of Pennsylvania. Environmental Pollution **149**:216-226.

Macedo-Sousa, J. A., J. L. T. Pestana, A. Gerhardt, A. J. A. Nogueira, and A. M. V. M. Soares. 2007. Behavioural and feeding responses of *Echinogammarus meridionalis* (Crustacea, Amphipoda) to acid mine drainage. Chemosphere **67**:1663-1670.

MacKay, R. J. 1992. Colonization by lotic macroinvertebrates: a review of processes and patterns. Canadian Journal of Fisheries and Aquatic Sciences **49**:617-628.

MacNeale, K. H., B. L. Peckarsky, and G. E. Likens. 2004. Contradictory results from different methods for measuring direction of insect flight. Freshwater Biology **49**:1260-1268.

MacNeale, K. H., B. L. Peckarsky, and G. E. Likens. 2005. Stable isotopes identify dispersal patterns of stonefly populations living along stream corridors. Freshwater Biology **50**:1117-1130.

Magurran, A. E. 1988. Ecological diversity and its measurement. Princeton University Press, Princeton, New Jersey.

Malmqvist, B. 2002. Aquatic invertebrates in riverine landscapes. Freshwater Biology **47**:679-694.

Malmqvist, B. and P. Hoffsten. 1999. Influence of drainage from old mine deposits on benthic macroinvertebrate communities in central Swedish streams. Water Research **33**:2415-2423.

Maret, T. R., D. J. Cain, D. E. MacCoy, and T. M. Short. 2003. Response of benthic invertebrate assemblages to metal exposure and bioaccumulation associated with hard-rock mining in northwestern streams, USA. Journal of the North American Benthological Society **22**:598-620.

Masters, Z., I. Peteresen, A. G. Hildrew, and S. J. Ormerod. 2007. Insect dispersal does not limit the biological recovery of streams from acidification. Aquatic conservation: Marine and Freshwater ecosystems **17**:375-383.

Matthaei, C. D., K. A. Peacock, and C. R. Townsend. 1999. Patchy surface stone movement during disturbance in a New Zealand stream and its potential significance for the fauna. Limnology and Oceanography **44**:1091-1102.

Matthaei, C. D., U. Uehlinger, E. I. Meyer, and A. Frutiger. 1996. Recolonization by benthic invertebrates after experimental disturbance in a Swiss prealpine river. Freshwater Biology **35**:233-248.

Matthaei, C. D., F. Weller, D. W. Kelly, and C. R. Townsend. 2006. Impacts of fine sediment addition to tussock, pasture, dairy and deer farming streams in New Zealand. Freshwater Biology **51**:2154-2172.

Matthaei, C. D., D. Werthmuller, and A. Frutiger. 1997. Invertebrate recovery from a bed-moving spate: the role of drift versus movements inside or over the substratum. Archiv für Hydrobiologie **140**:221-235.

Maurer, M. A. and M. A. Brusven. 1983. Insect abundance and colonisation rate in *Fontinalis neo-mexicana* (Bryophyta) in an Idaho batholith stream, U.S.A. Hydrobiologia **98**:9-15.

May, R. M., 1975. Patterns of species abundance and diversity Belknap Press of Harvard University, Cambridge, Massachusetts, U.S.A.

McClurg, S. E., T. J. Petty, P. M. Mazik, and J. L. Clayton. 2007. Stream ecosystem response to limestone treatment in acid impacted watersheds of the Allegheny Plateau. Ecological Applications **17**:1087-1104.

McCune, B. and J. Mefford. 1999. PC-ORD. Multivariate analysis of ecological data. MjM Software, Gleneden Beach, Oregon, U.S.A.

McGrath, K. E., E. T. H. M. Peeters, J. A. J. Beijer, and M. Scheffer. 2007. Habitat mediated cannibalism and macroinvertebrate restriction in the stream invertebrate *Gammarus pulex*. Hydrobiologia **589**:155-164.

McIntosh, A. R. 2002. Impact of introduced trout on *Nesameletus* mayfly populations measured using electrofishing. Archiv für Hydrobiologie **154**:375-392.

McIntosh, A. R., H. S. Greig, S. A. McMurtrie, P. Nystrom, and M. J. Winterbourn. 2005. Top-down and bottom-up influences on populations of a stream detritivore. Freshwater Biology **50**:1206-1218.

McMurtrie, S. 2000. Interactive effects of disturbance, food supply and predators on *Zelandopsyche ingens* (Trichoptera). MSc. University of Canterbury, Christchurch.

McMurtry, M. J. 1984. Avoidance of sublethal doses of copper and zinc by tubificid oligochaetes. Journal of Great Lakes Research **10**:267-272.

MED. 2007. New Zealand Energy Strategy to 2050. Ministry of Economic Development, Wellington.

MED. 2008. Annual Report- 2007-2008. Ministry of Economic Development (Crown Minerals), Wellington.

MED. 2009. Overview of the New Zealand Coal Industry. Ministry of Economic Development, Wellington.

Memon, P. A. and H. C. Perkins. 2000. Environmental planning and management: the broad context. Pages 10-31 *in* P. A. Memon and H. C. Perkins, editors. Environmental planning and management in New Zealand. Dunmore Press, Palmerston North.

Merrett, W. J., G. P. Rutt, N. S. Weatherley, S. P. Thomas, and S. J. Ormerod. 1991. The response of macroinvertebrates to low pH and increased aluminium concentration in Welsh streams: multiple episodes and chronic exposure. Archiv für Hydrobiologie **121**:115-125.

MfE. 2006. Getting in on the Act: an every day guide to the Resource Management Act. Ministry for Environment, Wellington.

Milisa, M., I. Habdija, B. Primc-Hadbija, and I. Radanovic. 2006. The role of flow velocity in the vertical distribution of particulate organic matter on moss-covered travertine barriers of the Plitvice Lakes (Croatia). Hydrobiologia **553**:231-243.

Miller, M. P., D. W. Blinn, and P. Keim. 2002. Correlations between observed dispersal capabilities and patterns of genetic differentiation in populations of four aquatic insect species from the Arizona White Mountains, U.S.A. Freshwater Biology **47**:1660-1673.

Milner, A. M. and I. T. Gloyne-Philips. 2005. The role of riparian vegetation and woody debris in the development of macroinvertebrate assemblages in streams. River Research and Applications **21**:403-420.

Milner, A. M., E. E. Knudsen, C. Soiseth, A. L. Robertson, D. Schell, I. T. Phillips, and K. Magnusson. 2000. Colonization and development of stream communities across a 200-year gradient in Glacier Bay National Park, Alaska, U.S.A. Canadian Journal of Fisheries and Aquatic Sciences **57**:2319-2335.

Mochizuki, S., Y. Kayaba, and K. Tanida. 2006. Drift patterns of particulate matter and organisms during artificial high flows in a large experimental channel Limnology and Oceanography **7**:93-102.

Mol, J. H. and P. E. Ouboter. 2004. Downstream effects of erosion from small-scale gold mining on the instream habitat and fish community of a small neotropical rainforest stream. Conservation Biology **18**:201-214.

Molinos, J. G. and I. Donohue. 2009. Differential contribution of concentration and exposure time to sediment dose effects on stream biota. Journal of the North American Benthological Society **28**:110-121.

Monteith, D. T., A. G. Hildrew, R. J. Flower, P. J. Raven, W. R. B. Beaumont, P. Collen, A. M. Kreiser, E. M. Shilland, and J. H. Winterbottom. 2005. Biological responses to the chemical recovery of acidified fresh waters in the UK. Environmental Pollution **137**:83-101.

Mossop, B. and M. J. Bradford. 2004. Importance of large woody debris for juvenile chinook salmon habitat in small boreal forest streams in the upper Yukon River basin, Canada. Canadian Journal of Forest Research **34**:1955-1966.

Müller, K. 1954. Investigations on the organic drift in North Swedish streams. Institute for Freshwater Research, Report **34**:133-148.

Müller, K. 1982. The colonization cycle of freshwater insects. Oecologia **52**:202-207.

Muotka, T. and P. Laasonen. 2002. Ecosystem recovery in restored headwater streams: the role of enhanced leaf retention. Journal of Applied Ecology **39**:145-156.

Muotka, T., R. Paavola, A. Haapala, M. Novikmec, and P. Laasonen. 2002. Long-term recovery of stream habitat structure and benthic invertebrate communities from in-stream restoration. Biological Conservation **105**:243-253.

Muotka, T. and J. Syrjanen. 2007. Changes in habitat structure, benthic invertebrate density, trout populations and ecosystem processes in restored forest streams: a boreal perspective. Freshwater Biology **52**:724-737.

Mykrä, H., J. Heino, and T. Muotka. 2007. Scale-related patterns in the spatial and environmental components of stream macroinvertebrate assemblage variation. Global Ecology and Biogeography **16**:149-159.

Nakano, S., H. Miyasaka, and N. Kuhara. 1999. Terrestrial-aquatic linkages: riparian arthropod inputs alter trophic cascades in a stream food web. Ecology **80**:2435-2441.

Nathan, S. 1978. Sheet 44, Greymouth, Geological map of New Zealand. New Zealand Department of Scientific and Industrial Research, Wellington.

Negishi, J. N. and J. S. Richardson. 2003. Responses of organic matter and macroinvertebrates to placements of boulder clusters in a small stream of southwestern British Columbia, Canada. Canadian Journal of Fisheries and Aquatic Sciences **60**:247-258.

Niyogi, D. K., W. M. Lewis, and D. M. McKnight. 2001. Litter breakdown in mountain streams affected by mine drainage: biotic mediation of abiotic controls. Ecological Applications **11**:506-516.

Niyogi, D. K., W. M. Lewis, and D. M. McKnight. 2002. Effects of stress from mine drainage on diversity, biomass and function in primary producers in mountain streams. Ecosystems **5**:554-567.

O'Halloran, K., J. Cavanagh, and J. S. Harding. 2008. Response of a New Zealand mayfly (*Deleatidium* spp.) to acid mine drainage: implications for mine remediation. Environmental Toxicology and Chemistry **27**:1135-1140.

O'Sullivan, A. 2005. Passive treatment technologies for managing metal mine wastes: lessons learnt from global applications Pages 279-300 *in* T. A. Moore, A. Black, J. A. Centeno, J. S. Harding, and D. A. Trumm, editors. Metal contaminants in New Zealand. Resolutionz Press, Christchurch.

Paetzold, A. and K. Tockner. 2005. Effects of riparian arthropod predation on the biomass and abundance of aquatic insect emergence. Journal of the North American Benthological Society **24**:395-402.

Palmer, M. A., J. D. Allan, and C. A. Butman. 1996. Dispersal as a regional process affecting the local dynamics of marine and stream benthic invertebrates. Trends in Ecology and Evolution **11**:322-326.

Palmer, M. A., R. F. Ambrose, and N. L. Poff. 1997. Ecological theory and community restoration ecology. Restoration Ecology **5**:291-300.

Paltridge, R. M., P. L. Dostine, C. L. Humphrey, and A. J. Boulton. 1997. Macroinvertebrate recolonisation after re-wetting of a tropical seasonally-flowing stream (Magela Creek, Northern Territory, Australia). Marine and Freshwater Research **48**:633-645.

Parkhill, K. L. and J. S. Golliver. 2002. Effect of inorganic sediment on whole stream productivity. Hydrobiologia **472**.

Peckarsky, B. L., B. W. Taylor, and C. C. Caudill. 2000. Hydrologic and behavioural constraints on oviposition of stream insects: implications for adult dispersal. Oecologia **125**:186-200.

Petersen, I., Z. Masters, A. G. Hildrew, and S. J. Ormerod. 2004. Dispersal of adult aquatic insects in catchments of differing land use. Journal of Applied Ecology **41**:934-950.

Petersen, I., J. H. Winterbottom, S. Orton, N. Friberg, A. G. Hildrew, D. C. Spiers, and W. S. C. Gurney. 1999. Emergence and lateral dispersal of adult Plecoptera and Trichoptera from Broadstone Stream, U.K. Freshwater Biology **42**:401-416.

Petrin, Z., G. Englund, and B. Malmqvist. 2008. Contrasting effects of anthropogenic and natural acidity in streams: a meta-analysis. Proceedings of the Royal Society B **275**:1143-1148.

Petrin, Z., H. Laudon, and B. Malmqvist. 2007a. Does freshwater macroinvertebrate diversity along a pH-gradient reflect adaptation to low pH? Freshwater Biology **52**:2172-2183.

Petrin, Z., B. McKie, I. Buffman, H. Lauden, and B. Malmqvist. 2007b. Landscape controlled chemistry variation affects communities and ecosystem function in headwater streams. Canadian Journal of Fisheries and Aquatic Sciences **64**:1563-1572.

Pfankuck, D. J. 1975. Stream reach inventory and channel stability evaluation. USDA Forest Service, Northern Region, Montana.

Pond, G. J., M. E. Passmore, F. A. Borsuk, L. Reynolds, and C. J. Rose. 2008. Downstream effects of mountaintop coal mining: comparing biological conditions using family- and genus-level macroinvertebrate bioassessment tools. Journal of the North American Benthological Society **27**:717-737.

Pretty, J. L. and M. Dobson. 2004. The response of macroinvertebrates to artificially enhanced detritus levels in plantation streams. Hydrology and Earth System Sciences **8**:550-559.

Quinn, J. M., A. B. Cooper, R. J. Davies-Colley, J. C. Rutherford, and R. B. Williamson. 1997. Land-use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill country streams. New Zealand Journal of Marine and Freshwater Research **31**:579-597.

Quinn, J. M., R. J. Davies-Colley, C. W. Hickey, M. L. Vickers, and P. A. Ryan. 1992. Effects of clay discharge on streams 2: benthic invertebrates Hydrobiologia **248**:235-247.

Quinn, J. M. and C. W. Hickey. 1990. Magnitude of effects of substrate particle size, recent flooding and catchment development on benthic invertebrates in 88 New Zealand rivers. New Zealand Journal of Marine and Freshwater Research **24**:411-427.

Quinn, J. M., N. R. Phillips, and S. M. Parkyn. 2007. Factors influencing retention of coarse particulate organic matter in streams. Earth Surface Processes and Landforms **32**:1186-1203.

Raddum, G. G. and B. L. Skjelkvale. 1995. Critical limits of acidification to invertebrates in different regions of Europe. Water, Air, and Soil Pollution **85**:475-480.

Raddum, G. G. and B. L. Skjelkvale. 2001. Critical limit of acidifying compounds to invertebrates in different regions of Europe. Water, Air, and Soil Pollution **130**:825-830.

Reeves, P., K. J. Collier, and A. M. Suren. 2004. Aquatic and riparian vegetation of rivers and streams. Pages 14.01-14.16 *in* J. S. Harding, P. Mosley, C. Pearson, and B. Sorrell, editors. Freshwaters of New Zealand. New Zealand Limnological Society, Christchurch.

Reich, P. and B. J. Downes. 2003. The distribution of aquatic invertebrate egg masses in relation to physical characteristics of oviposition sites at two Victorian upland streams. Freshwater Biology **48**:1497-1513.

Resh, V. H. 2005. Stream crossings and the conservation of diadromous invertebrates in South Pacific island streams. Aquatic conservation: Marine and Freshwater ecosystems **15**:313-317.

Richardson, J. S. and P. M. Kiffney. 2000. Responses of a macroinvertebrate community from a pristine, Southern British Columbia, Canada, stream to metals in experimental mesocosms. Environmental Toxicology and Chemistry **19**:736-743.

Rounick, J. S. and M. J. Winterbourn. 1983. The formation, structure and utilization of stone surface organic layers in two New Zealand streams. Freshwater Biology **13**:57-72.

Rowe, D. K. and J. Smith. 2003. Use of in-stream cover types by adult banded kokopu (*Galaxias fasciatus*) in first-order North Island, New Zealand, streams. New Zealand Journal of Marine and Freshwater Research **37**:541-552.

Ruetz Iii, C. R., R. M. Newman, and B. Vondracek. 2002. Top-down control in a detritus-based food web: fish, shredders, and leaf breakdown. Oecologia **132**:307-315.

Ryan, P. A. 1991. Environmental effects on sedimentation on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research **25**:207-221.

Sanderson, R. A., M. D. Eyre, and S. P. Rushton. 2005. The influence of stream invertebrate composition at neighbouring sites on local assemblage composition. Freshwater Biology **50**:221-231.

Sarriquet, P. E., P. Bordenave, and P. Marmonier. 2007. Effects of bottom sediment restoration on interstitial habitat characteristics and benthic macroinvertebrate assemblages in a headwater stream. River Research and Applications **23**:815-828.

Scarsbrook, M. R. 2000. Life-histories. Pages 76-99 *in* K. J. Collier and M. J. Winterbourn, editors. New Zealand stream invertebrates: Ecology and implications for management. The Caxton Press, Christchurch.

Scarsbrook, M. R. and J. Halliday. 2002. Detecting patterns in hyporheic community structure: Does sampling method alter the story? New Zealand Journal of Marine and Freshwater Research **36**:443-453.

Scealy, J. A., S. J. Mika, and A. J. Boulton. 2007. Aquatic macroinvertebrate communities on wood in an Australian lowland river: experimental assessment of the interactions of habitat, substrate complexity and retained organic matter. Marine and Freshwater Research **58**:153-165.

Schlief, J. 2004. Leaf associated microbial activities in a stream affected by acid mine drainage. International Review of Hydrobiology **89**:467-475.

Schlief, J. and M. Mutz. 2006. Palatability of leaves conditioned in streams affected by mine drainage: a feeding experiment with *Gammarus pulex* (L.). Hydrobiologia **563**:445-452.

Schneider, K. N. and K. O. Winemiller. 2008. Structural complexity of woody debris patches influences fish and macroinvertebrate species richness in a temperate floodplain-river system. Hydrobiologia **610**:235-244.

Schorr, M. S. and J. C. Backer. 2006. Localised effects of coal mine drainage on fish assemblages in a Gumberland Plateau stream in Tennesse. Journal of Freshwater Ecology **21**:17-24.

Schultheis, A. S. and J. M. Hughes. 2005. Spatial patterns of genetic structure among populations of a stone-cased caddis (Trichoptera: Tasimiidae) in south-east Queensland, Australia. Freshwater Biology **50**:2002-2010.

Schultheis, A. S., M. Sanchez, and A. C. Hendricks. 1997. Structural and functional responses of stream insects to copper pollution. Hydrobiologia **346**:85-93.

Shaw, E. A. and J. S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mylass*) growth and survival. Canadian Journal of Fisheries and Aquatic Sciences **58**:2213-2221.

Shearer, K. A., J. D. Stark, J. W. Hayes, and G. W. Young. 2003. Relationships between drifting and benthic invertebrates in 3 New Zealand Rivers: implications for drift-feeding fish. New Zealand Journal of Marine and Freshwater Research **37**:809-820.

Sheldon, A. L. and M. L. Warren Jr. 2009. Filters and templates: Stonefly (Plecoptera) richness in Ouachita Mountains streams, U.S.A. Freshwater Biology **54**:943-956.

Small, M. J., M. W. Doyle, R. L. Fuller, and R. B. Manners. 2008. Hydrologic versus geomorphic limitation on CPOM storage in stream ecosystems. Freshwater Biology **53**:1618-1631.

Smith, B. J. 2001. Larval Hydrobiosidae. Pages 1-52 *in* B. J. Smith, editor. Biodiversity identification workshop. National Institute of Water and Atmospheric Research, Christchurch, New Zealand.

Sode, A. and P. Wiberg-Larsenm. 1993. Dispersal of adult Trichoptera at a Danish forest brook. Freshwater Biology **30**:439-446.

Soucek, D. J., D. S. Cherry, and C. E. Zipper. 2003. Impacts of mine drainage and other nonpoint source pollutants on aquatic biota in the upper Powell River system, Virginia. Human and Ecological Risk Assessment **9**:1059-1073.

Soucek, D. J., B. C. Denson, T. S. Schmidt, D. S. Cherry, and C. E. Zipper. 2002. Impaired *Acroneuria* sp. (Plecoptera, Perlidae) populations associated with aluminium contamination in neutral pH surface waters. Archives of Environmental Contamination and Toxicology **42**:416-422.

Stark, J. D. 1985. A macroinvertebrate community index of water quality for stoney streams. Ministry of Works and Development, Wellington.

Stark, J. D. 1993. Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity and substratum on index values. New Zealand Journal of Marine and Freshwater Research **27**:463-478.

Stark, J. D. 1998. SQMCI: a biotic index for freshwater macroinvertebrate coded-abundance data. New Zealand Journal of Marine and Freshwater Research **32**:55-66.

Stark, J. D., I. K. G. Boothroyd, J. S. Harding, J. R. Maxted, and M. R. Scarsbrook. 2001. Protocols for sampling macroinvertebrates in wadeable streams. Ministry for Environment, Wellington, New Zealand.

StatSoft. 2008. Statistica. Tulsa, Oklahoma.

Stephenson, S. L., S. M. Studlar, C. J. McQuattie, and P. J. Edwards. 1995. Effects of acidification on bryophyte communities in West Virginia mountain streams. Journal of Environmental Quality **24**:116-125.

Suren, A., S. McMurtie, and L. O'Brien. 2004. River restoration. Pages 38.01-38.20 *in* J. Harding, P. Mosley, C. Pearson, and B. Sorrell, editors. Freshwaters of New Zealand. New Zealand Hydrological Society Inc. and New Zealand Limnological Society Inc., Christchurch.

Suren, A. M. 1990. The ecological role of bryophytes in alpine streams of New Zealand. PhD. University of Canterbury, Christchurch.

Suren, A. M. 1991. Bryophytes as invertebrate habitat in two New Zealand alpine streams. Freshwater Biology **26**:399-418.

Suren, A. M. and M. J. Duncan. 1999. Rolling stones and mosses: effect of substrate stability on bryophyte communities in streams. Journal of the North American Benthological Society **18**:457-467.

Suren, A. M. and I. G. Jowett. 2001. Effects of deposited sediment on invertebrate drift: an experimental study. New Zealand Journal of Marine and Freshwater Research **35**:725-737.

Suren, A. M., M. L. Martin, and B. J. Smith. 2005. Short-term effects of high suspended sediments on six common New Zealand stream invertebrates. Hydrobiologia **548**:67-74.

Suren, A. M. and M. J. Winterbourn. 1991. Consumption of aquatic bryophytes by alpine stream invertebrates in New Zealand. New Zealand Journal of Marine and Freshwater Research **25**:331-343.

Suren, A. M. and M. J. Winterbourn. 1992. The influence of periphyton, detritus and shelter on invertebrate colonization of aquatic bryophytes. Freshwater Biology **27**:327-339.

Sutcliffe, D. W. and A. G. Hildrew, editors. 1989. Invertebrate communities in acid streams. Cambridge University Press, Cambridge, New York.

Sweka, J. A. and K. J. Hartman. 2001. Influence of turbidity on brook trout reactive distance and foraging success. Transactions of the North American Fisheries Society **130**:138-146.

Theinhardt, N. A. 2003. Plant restoration at an open-cast coal mine, West Coast, New Zealand. MSc. University of Canterbury, Christchurch.

Thompson, R. M. and C. R. Townsend. 2000. New Zealand's stream invertebrate communities: an international perspective. Pages 53-74 *in* K. J. Collier and M. J. Winterbourn, editors. New Zealand stream invertebrates: ecology and implications for management. New Zealand Limnological Society, Christchurch.

Tikkanen, P., P. Lassonen, T. Muotka, A. Huhta, and K. Kuusela. 1994. Short-term recovery of benthos following disturbance from stream habitat rehabilitation. Hydrobiologia **273**:121-130.

Townsend, C. R. and A. G. Hildrew. 1976. Field experiments on drifting, colonization and continuous redistribution of stream benthos. Journal of Animal Ecology **45**:759-772.

Townsend, C. R., S. S. Uhlmann, and C. D. Matthaei. 2008. Individual and combined responses of stream ecosystems to multiple stressors. Journal of Applied Ecology **45**:1810-1819.

Turner, D. and D. D. Williams. 2000. Invertebrate movements within a small stream: Density dependence or compensating for drift? International Review of Hydrobiology **85**:141-156.

Urban, M. C. 2004. Disturbance heterogeneity determines freshwater metacommunity structure. Ecology **85**:2971-2978.

Van Damme, P. A., C. Hamel, A. Ayala, and L. Bervoets. 2008. Macroinvertebrate community response to acid mine drainage in rivers of the High Andes (Bolivia). Environmental Pollution **156**:1061-1068.

Vanschoenwinkel, B., S. Gielen, M. Seaman, and L. Brendonck. 2008. Any way the wind blows - Frequent wind dispersal drives species sorting in ephemeral aquatic communities. Oikos **117**:125-134.

Verb, R. G. and M. L. Vis. 2000. Comparison of benthic diatom assemblages from streams draining abandoned and reclaimed coal mines and nonimpacted sites. Journal of the North American Benthological Society **19**:274-288.

Wagner, F. 2003. Flight behaviour of merolimnic insects from the Leutra River (Thuringia, Germany) Aquatic Insects **25**:51-62.

Wallace, J. B., S. L. Eggert, J. L. Meyer, and J. R. Webster. 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. Science **277**:102-104.

Wallace, J. B., S. L. Eggert, J. L. Meyer, and J. R. Webster. 1999. Effects of resource limitation on a detrital-based ecosystem. Ecological Monographs **69**:409-442.

Wallace, J. B., A. D. Huryn, and G. J. Lugthart. 1991. Colonization of a headwater stream during three years of seasonal insecticidal applications. Hydrobiologia **211**:65-76.

Wallace, J. B., D. S. Vogel, and T. F. Cuffney. 1986. Recovery of a headwater stream from an insecticide-induced community disturbance. Journal of the North American Benthological Society **5**:115-126.

Wallace, J. B. and J. R. Webster. 1996. The role of macroinvertebrates in stream ecosystem function. Annual review of Entomology **41**:115-139.

Wantanabe, N. C., S. Harada, and Y. Komai. 2000. Long-term recovery from mine drainage disturbance of a macroinvertebrate community in the Ichi-kawa River, Japan. Hydrobiologia **429**:171-180.

Waringer, J. A. 1991. Phenology and the influence of meteorological parameters on the catching success of light-trapping for Trichoptera. Freshwater Biology **25**:307-319.

Watton, A. J. and H. A. Hawkes. 1984. The acute toxicology of ammonia and copper to the gastropod *Potamopyrgus jenkinsi* (Smith). Environmental Pollution Series A **36**:17-29.

Watts, C. H., M. Vojvodic-Vukovic, G. C. Arnold, and R. K. Didham. 2008. A comparison of restoration techniques to accelerate recovery of litter decomposition and microbial activity in an experimental peat bog restoration trial. Wetlands Ecology and Management **16**:199-217.

Watzen, K. M. 2006. Physical pollution: effects of gully erosion on benthic invertebrates in a tropical clear-water stream. Aquatic conservation: Marine and Freshwater ecosystems **16**:733-749.

Weber, P. A., W. M. Skinner, J. B. Hughes, P. Lindsay, and T. A. Moore. 2006. Sources of Ni in coal mine acid rock drainage, West Coast, New Zealand. International Review of Coal Geology **67**:214-220.

Webster, J. R. and E. F. Benfield. 1986. Vascular plant break-down in freshwater ecosystems. Annual Review of Ecology and Systematics **17**:567-594.

Weis, J. S. and P. Weis. 1989. Tolerance and stress in a polluted environment: the case of the mummichog. Bioscience **39**:89-95.

Wellnitz, T. A. and S. P. Sheldon. 1995. The effects of iron and manganese on diatom colonization in a Vermont stream. Freshwater Biology **34**:465-470.

White, D. S. 1993. Perspectives on defining and delineating hyporheic zones. Journal of the North American Benthological Society **12**:61-69.

Wilcock, H. R., M. W. Bruford, R. A. Nichols, and A. G. Hildrew. 2007. Landscape, habitat characteristics and the genetic population structure of two caddisflies. Freshwater Biology **52**:1907-1929.

Wilcock, H. R., R. A. Nichols, and A. G. Hildrew. 2003. Genetic population structure and neighbourhood population size estimates of the caddisfly *Plectrocnemia conspersa*. Freshwater Biology **48**:1813-1824.

Williams, D. D. 1996. Environmental constraints in temporary fresh waters and their consequences for the insect fauna Journal of the North American Benthological Society **15**:634-650.

Williams, D. D. and H. B. N. Hynes. 1976. Recolonisation mechanisms of stream benthos. Oikos **27**:265-272.

Winterbourn, M. J. 1982. The invertebrate fauna of a forest stream and its association with fine particulate matter. New Zealand Journal of Marine and Freshwater Research **16**:271-281.

Winterbourn, M. J. 1998. Insect faunas of acidic coal mine drainages in Westland, New Zealand. New Zealand Entomologist **21**:65-72.

Winterbourn, M. J. 2004. Stream invertebrates. Pages 16.01-16.14 *in* J. S. Harding, P. Mosley, C. Pearson, and B. Sorrell, editors. Freshwaters of New Zealand. New Zealand Hydrological Society Inc. and New Zealand Limnological Society Inc., Christchurch.

Winterbourn, M. J. 2005. Dispersal, feeding and parasitism of adult stoneflies (Plecoptera) at a New Zealand forest stream. Aquatic Insects **27**:155-166.

Winterbourn, M. J. and K. J. Collier. 1987. Distribution of benthic invertebrates in acid, brown water streams in the South Island of New Zealand. Hydrobiologia **153**:277-286.

Winterbourn, M. J. and A. L. M. Crowe. 2001. Flight activity of insects along a mountain stream: is directional flight adaptive? Freshwater Biology **46**:1479-1489.

Winterbourn, M. J., K. L. D. Gregson, and C. H. Dolphin. 2000b. Guide to the aquatic insects of New Zealand. 13 edition. Bulletin of the Entomological Society of New Zealand.

Winterbourn, M. J., J. S. Harding, and A. R. McIntosh. 2007. Response of the benthic fauna of an urban stream during six years of restoration. New Zealand Natural Sciences **32**:1-12.

Winterbourn, M. J. and W. F. McDiffett. 1996. Benthic faunas of streams of low pH but contrasting water chemistry in New Zealand. Hydrobiologia **341**:101-111.

Winterbourn, M. J., W. F. McDiffett, and S. J. Eppley. 2000a. Aluminium and iron burdens of aquatic biota in New Zealand streams contaminated by acid mine drainage: effects of trophic level. The Science of the Total Environment **254**:45-54.

Winterbourn, M. J., J. S. Rounick, and B. Cowie. 1981. Are New Zealand stream ecosystems really that different? New Zealand Journal of Marine and Freshwater Research **15**:231-328.

Wipfli, M. S., J. S. Richardson, and R. J. Naiman. 2007. Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. Journal of the American Water Resources Association **43**:72-85.

Wolfe, J. E. and D. T. Lind. 2008. Influence of suspended clay on phosphorus uptake by periphyton. Hydrobiologia **610**:211-222.

Wood, P. J. and P. D. Armitage. 1997. Biological effects of fine sediment in the lotic environment. Environmental Management **21**:203-217.

Yamada, H. and F. Nakamura. 2002. Effect of fine sediment deposition and channel works on periphyton biomass in the Makomanai River, northern Japan. River Research and Applications **18**:481-493.

Zbinden, M., M. Hieber, C. T. Robinson, and U. Uehlinger. 2008. Short-term colonization patterns of macroinvertebrates in alpine streams. Fundamental and Applied Limnology **171**:75-86.

Zickovich, J. M. and A. J. Bohonak. 2007. Dispersal ability and genetic structure in aquatic invertebrates: a comparative study in southern California streams and reservoirs. Freshwater Biology **52**:1982-1996.

Appendix 1

Summary physical, chemical and biological results for each of the 45 streams sampled on the Stockton Plateau during a single occasion between December 2007- January 2008.

Stream	Impact category	Easting	Northing	pH	EC	Turbidity (NTU)	Diss. Fe	Diss. Al.	Diss. Zn	Diss. Ni	Substrate	Width	Depth	Total taxa	% EPT	Total density
Stream 1	R	2414833	5944935	5.6	58.6	0.27	0.11	0.25	0.002	0.0005	Bryophytes	0.4	0.12	0	0	0
Stream 2	R	2414833	5945258	5.8	24.3	0.62	0.21	0.16	0.003	0.0005	Bryophytes	1.2	0.06	12	58	383.3
Repo	R	2419929	5952937	5.5	32.5	1.50	0.27	0.30	0.005	0.0005	Bedrock	10.0	0.09	14	64	2771.7
MT6	R	2419609	5952750	4.1	44.9	0.27	0.04	0.14	0.010	0.0074	Bryophytes	3.2	0.10	13	38	721.7
MT6a	R	2419613	5952746	4.1	49.5	0.21	0.03	0.26	0.029	0.0015	Bedrock	1.0	0.01	6	16	316.7
M2	R	2416357	5950327	3.2	116	0.34	0.10	0.29	0.006	0.0006	Bryophytes	1.1	0.15	7	43	126.7
Twins	R	2413983	5951522	6.6	124	1.10	0.30	0.06	0.004	0.0001	Cobble	2.8	0.20	12	42	226.7
Bradley	R	2415235	5953413	5.2	164	1.60	0.75	0.87	0.014	0.0053	Cobble	1.4	0.10	6	33	58.3
Jones	R	2411510	5947832	7.6	56.5	0.19	0.04	0.12	0.002	0.0005	Cobble	7.0	0.30	13	62	2830.0
Kerr	R	2411529	5947874	7.6	88.6	0.15	0.02	0.04	0.002	0.0005	Cobble	3.0	0.20	16	69	449.3
Whorley	R	2412572	5949256	7.7	56.5	0.14	0.03	0.08	0.002	0.0005	Cobble	3.0	0.30	19	63	3704.3
Random 2	R	2418414	5949948	4.2	25.2	0.66	0.20	0.20	0.002	0.0005	Bedrock	1.7	0.01	7	43	121.7
MCT 2	R	2417533	5951688	3.8	66.6	2.20	0.55	0.28	0.014	0.0031	Bedrock	0.5	0.05	2	50	16.7

Rudolph	R			6.3	158	0.50	0.02	0.04	0.002	0.0005	Cobble	2.3	0.07	26	65	2711.0
T69	R	2418447	5948904	5.7	24.9	1.30	0.15	0.10	0.004	0.0005	Bedrock	1.8	0.02	15	47	660.0
T35	R	2418604	5948607	4.4	23.5	0.50	0.14	0.32	0.009	0.0014	Bedrock	1.5	0.01	23	61	1841.0
M1	R	2416881	5950508	3.1	318	1.10	2.10	2.40	0.082	0.0038	Bedrock	2.3	0.05	11	45	250.0
Hot	I	2416055	5951590	2.5	1617	0.37	19.00	49.00	0.550	0.0970	Bedrock	0.5	0.3	2	0	23.3
Mine Portal	I	2415963	5951543	2.7	1156	0.40	11.00	45.00	0.470	0.0830	Bedrock	0.3	0.02	3	0	1150.0
Ford	I	2417462	5949135	2.5	944	3.40	15.00	42.00	0.920	0.1900	Bedrock	4.3	0.02	2	33	133.3
Miller Ck	I	2416428	5951744	2.7	851	0.63	8.10	23.00	0.320	0.0490	Bedrock	7.4	0.10	3	0	68.7
Fly Ck Main	I	2416851	5946894	3.3	505	0.68	1.90	33.00	0.640	0.1300	Bedrock	4.8	0.20	0	33	10.0
Fly Ck 1	I	2416821	5946877	3.4	424	0.33	0.93	29.00	0.660	0.1300	Bedrock	1.6	0.13	3	33	100.0
Fly Ck 2	I	2416829	5946904	3.2	585	0.70	3.10	40.00	0.630	0.1400	Bedrock	2.9	0.15	4	25	38.0
Herbert	I	2415229	5944087	2.7	481	6.9	1.80	10.00	0.160	0.0410	Bedrock	1.0	0.90	9	56	133.3
Granity	I	2415728	5952020	3.5	154	5.6	1.00	3.50	0.075	0.0150	Bedrock	4.9	0.20	7	43	183.3
Plover	I	2415664	5945468	3.3	150	13	0.90	4.70	0.120	0.0280	Cobble	2.3	0.38	1	0	16.7
Plover 2	I	2416033	5945607	3.4	119	0.19	0.11	2.50	0.097	0.0220	Cobble	1.8	0.06	4	30	3055.0
WW4	I	2415368	5944437	3.3	363	0.54	0.97	13.00	0.200	0.0600	Bedrock	0.8	0.11	5	40	201.0
Sandy	I	2418739	5952332	2.8	385	25.00	8.70	16.00	0.290	0.0640	Cobble	1.7	0.07	2	0	58.3

MCT 1	I	2417629	5951782	2.6	407	6.2	5.10	6.50	0.140	0.0150	Bedrock	8.7	0.05	2	0	66.7
S16	I	2419061	5947285	3.3	278	0.22	1.00	8.20	0.190	0.0330	Bedrock	12.5	0.10	5	40	350.0
Pack	I	2416195	5951658	2.6	1301	0.63	11	30.00	0.410	0.0690	Root mat	0.8	0.02	1	0	4316.7
M3	I	2416354	5949137	2.6	1314	3.60	58	80.00	2.800	0.5000	Cobble	3.5	0.06	1	0	8.3
S6	I	2417412	5952236	2.9	531	2.80	1.20	1.80	0.044	0.0067	Bedrock	6.9	0.03	4	25	2058.3
S23	I	2417036	5952205	2.6	1405	1.00	5.80	22.00	0.320	0.0590	Bedrock	7.8	0.03	1	0	116.7
S14	I	2417794	5949722	6.0	865	304	1.8	0.05	0.780	0.2400	Bedrock	6.6	0.02	0	0	0
WW1	MI	2415830	5944898	3.5	159	0.90	0.12	3.90	0.100	0.0220	Bryophyte	3.1	0.20	6	50	576.7
WW2	MI	2415950	5944757	3.3	165	0.18	0.10	4.20	0.110	0.0240	Bryophyte	1.6	0.20	6	22	338.3
WW3	MI	2416312	5945081	3.6	106	0.46	0.08	2.50	0.080	0.0170	Bryophyte	1.8	0.13	9	20	308.3
Plover 3	MI	2415902	5946065	3.3	557	1.4	1.5	43.00	0.800	0.1500	Cobble	2.3	0.06	10	38	402.0
Weka	MI	2418893	5952435	3.7	58.1	0.76	0.31	0.60	0.019	0.0031	Silt	3.5	0.3	8	0	38.0
Random 1	MI	2418119	5949712	3.8	39.1	1.00	0.29	0.18	0.008	0.0020	Bedrock	2.0	0.01	4	43	0
T31	MI	2419027	5947540	3.3	262	0.16	1.1	1.40	0.026	0.0045	Cobble	3.5	0.02	7	100	43.0
Stream 3	MI	2414883	5945708	5.0	24.4	0.30	0.24	0.24	0.003	0.0005	Silt	1.7	0.05	2	0	100.0

Appendix 2

Summary of species and family densities (m^2) for benthic (B), drift (D) and F (downstream flight) from the six sampled streams. Macroinvertebrates from benthic and drift samples have been taken to family or species, while adult insects has been taken to family. Italicised numbers are for the densities for genus.

		Repo			MT6			Weka			T31			T35			T69			Ratio	Ratio
Family	Genus	B	D	F	B	D	F	B	D	F	B	D	F	B	D	F	B	D	F	B: D	B: F
Leptophlebiidae	*Austroclima*	*2267*	*1811*		*8*									*158*	*73*		*17*			2150: 1884	2450: 0
	Deleatidium	*42*	*65*		*92*	*58*			*60*			*5*		*45*	*73*		*45*	*36*		223: 297	223: 0
	Zephlebia													*42*	*3*					42: 3	42: 0
Coloburiscidae	*Coloburiscus*	*200*	*251*																	200: 251	200: 0
Austroperlidae	*Austroperla*	*17*	*17*		*58*	*8*								*367*	*3*					442: 29	442: 0
Gripopterygidae	*Zealandobius*	*42*							*4*									*4*		42: 8	42: 0
	Zelandoperla		*9*																	9: 0	9: 0
	Taraperla							*2*						*283*			*25*			310: 0	310: 0
	Megaleptoperla													*33*				*174*		50: 4	50: 0
Hydropsychidae	*Aoteapsyche*		*4*	0.2			0.3							*25*	*3*					25: 8	25: 0.5
Notonemouridae	*Spaniocercoides*										*25*									25: 0	25: 0
	Spaniocerca	*17*																		17: 0	17: 0
Hydrobiosidae	*Hydrobiosis*	*42*	*26*				0.2			0.1				*5*			*8*		0.1	55: 26	55: 0.4
	Psilochorema	*17*	*43*		*67*	*50*			*12*		*17*	*28*		*25*			*125*	*8*		250: 129	250: 0

Helicopsychidae	*Rakiura*										*8*			0.1	8: 0	8: 0.1
Helicophidae	*Zelolessica*										*8*				8: 0	8: 0
Hydroptilidae	*Oxyethira*	*33*	*4*		*33*		*267*	*204*	*283*	*555*	*42*	*27*	*36*		625: 859	625: 0
Oeconesidae	*Oeconesus*										*17*				17: 0	17: 0
Conoesucidae	*Pycnocentria*	*17*		*42*		0.1		*8*			*53*				120: 0	120:0.1
Ameletopsidae	*Ameletopsis*										*13*		*24*		0: 37	0: 0
Economidae	*Zelandoptila*		*26*												0: 26	0: 0