

Factors limiting invertebrate recovery during stream restoration

A thesis

submitted in partial fulfilment of the requirements for the

Degree of

Master of Science in Ecology

in the

University of Canterbury

by

Kimberley J. Roberts

University of Canterbury

2012

TABLE OF CONTENTS

FIGURES AND TABLES.....	4
ABSTRACT.....	6
CHAPTER ONE: General introduction	7
INTRODUCTION, pg. 8	
CASE STUDY, pg. 10	
<i>Stream from regime, pg. 15</i>	
<i>Species pools and habitat fragmentation, pg. 16</i>	
<i>Restoration priorities, pg. 18</i>	
THESIS STRUCTURE, pg. 19	
CHAPTER TWO: Factors limiting colonisation and recovery of sensitive stream insects during stream restoration.....	20
INTRODUCTION, pg. 21	
METHODS, pg. 24	
<i>Sites and in-stream habitat quality, pg. 24</i>	
<i>Larvae and adult population survey, pg. 25</i>	
<i>Oviposition experiment, pg. 26</i>	
<i>Statistical analysis, pg. 27</i>	
RESULTS, pg. 28	
<i>Habitat quality, pg. 28</i>	
<i>Community patterns, pg. 28</i>	
<i>Oviposition, pg. 36</i>	
DISCUSSION, pg. 37	
<i>Conclusions, pg. 44</i>	

CHAPTER THREE: Is active sediment removal an option when fine sediments limit invertebrate recovery in stream restoration?.....45

INTRODUCTION, pg. 46

METHODS, pg. 49

Rock-basket experimental setup, pg. 49

Sediment removal experimental setup, pg. 49

Statistical analysis, pg. 50

RESULTS, pg. 53

Rock-basket experiment, pg. 53

Sediment removal experiment, pg. 57

DISCUSSION, pg. 63

CHAPTER FOUR: Management recommendations.....69

APPLICATION OF THE FINDINGS AT RIVERSDALE STREAM, pg. 70

In this chapter, pg. 71

How can we improve restoration success? pg. 71

Determining limiting factors in stream restoration, pg. 74

Priorities and goal setting for stream restoration, pg. 78

Conclusions, pg. 80

ACKNOWLEDGEMENTS.....82

REFERENCES.....84

FIGURES AND TABLES

CHAPTER ONE

Figures

- Figure 1. Map of general study area, pg. 11
- Figure 2. Photograph of Riversdale Stream displaying eroded banks, pg. 12
- Figure 3. Photograph of Riversdale Stream displaying filamentous algal growth, pg. 13
- Figure 4. Photograph of Riversdale Stream displaying excessive sediment, pg. 14
- Figure 5. Photograph of Riversdale Stream displaying excessive macrophyte growth, pg. 14
- Figure 6. Graph of Riversdale Streams average stage height, pg. 15
- Figure 7. Photograph of Riversdale Stream displaying poor habitat during drought, pg. 16

CHAPTER TWO

Figures

- Figure 1. Map of study sites, pg. 25
- Figure 2. Average larvae richness at study sites across a habitat gradient, pg. 29
- Figure 3. Adult and larvae community and species composition ordinations from all sites, pg. 31
- Figure 4. Average abundance of adult Trichoptera from all sites, pg. 33
- Figure 5. Average and total adult and larvae richness from all sites, pg. 34
- Figure 6. Average abundance of Hydrobiosidae egg masses at Riversdale sites, pg. 37

Tables

- Table 1. Summary of adult Trichoptera family response to impact levels, pg. 32
- Table 2. Summary of adult and larvae Trichoptera family response to impact levels, pg. 35
- Table 3. Summary of the average abundance of egg masses at Riversdale Stream, pg. 36
- Table 4. Summary of habitat preferences of absent Trichoptera from Riversdale Stream, pg. 39

CHAPTER THREE

Figures

- Figure 1. Experimental design of sediment removal experiment at Riversdale Stream, pg. 51
- Figure 2. Community and species ordination of invertebrates from added and natural habitat nested within high and low sediment levels found at Riversdale Stream, pg. 54
- Figure 3. Community responses of invertebrate populations from added and natural habitat nested with high and low sediment levels found at Riversdale Stream, pg. 55
- Figure 4. Average sediment depth through time for sediment removal treatments, pg. 57
- Figure 5. Habitat variable changes from sediment removal experiment treatments, pg. 58
- Figure 6. Community and species composition ordination of invertebrate populations from sediment removal treatments at Riversdale Stream, pg. 60
- Figure 7. Community responses of invertebrate populations from sediment removal treatments at Riversdale Stream, pg. 61

Tables

- Table 1. Summary of community response variables of added and natural habitat, nested within high and low sediment levels found at Riversdale Stream, pg. 56
- Table 2. Summary of habitat variable change through time, pg. 59
- Table 3. Summary of community response variable's from sediment removal treatments, pg. 62

CHAPTER FOUR

Figures

- Figure 1. Factors to consider when choosing restoration target species, pg. 73
- Figure 2. Processes to determine if active sediment removal is an option, pg. 77
- Figure 3. Priorities for restoration management, pg 79

ABSTRACT

Many stream restoration projects, as part of returning a degraded ecosystem to a healthier state, aim to restore aquatic invertebrate populations. Unfortunately, many attempts only 'beautify' streams without achieving improvements in biodiversity. Lack of connectivity of a restoration site to a regional species pool may explain some failures. I tested this by collecting larval and adult aquatic insects from an agriculturally impacted Canterbury high country stream to evaluate connectivity of the regional species pool. The stream was surrounded by high-quality habitat in an adjacent National Park. Surrounding streams contained diverse assemblages of aquatic insects, but processes in the environment and limitations of in-stream habitat meant their adults did not always arrive at the target. In addition, oviposition habitat for hydrobiosid caddisflies was added to sections of stream and compared to un-manipulated control sections to test oviposition site limitation. The addition of oviposition habitat led to more hydrobiosid egg masses in comparison to control reaches. However, oviposition was also limited by in-stream habitat conditions, particularly the abundance of fine sediments. Sedimentation is a common pollutant in streams and is linked to decreases in habitat, food resources, and invertebrate populations. Moreover, common restoration methods, such as riparian management, have little success at reversing already high sediment levels, and are therefore insufficient to bring improvements to in-stream communities or sought-after habitat conditions. Therefore, after determining sediment was restricting sensitive invertebrate recovery at Riversdale Stream, by adding patches of high quality habitat I experimentally compared the factorial effects of sediment flushing and channel narrowing on sediment removal. Treatments improved habitat and prompted recovery of sensitive invertebrates, but an interactive effect where both flushing and channel narrowing combined created the most improved habitat conditions and the greatest improvements of invertebrate communities. Thus, while habitat improvements are an important part of restoration, features limiting species recovery such as connectivity and sedimentation, are particularly important.

CHAPTER ONE

General introduction



Waimakariri River valley from the Binser Saddle track, Arthur's Pass National Park

INTRODUCTION

Both stream and river communities are adversely affected by land-use change through a series of direct and indirect effects on habitat quality, resource availability, water chemistry and biotic interactions (Allan and Johnson 1997, Harding *et al.* 1998, Harding 2003, Allan 2004). With growing awareness of this degradation of natural habitat has come increased efforts to restore these environments. Recently, the desire to return biodiversity to streams and rivers on degraded landscapes together with the change in emphasis from single species to entire ecosystem restoration has emerged as a focus (Doppelt 1993, Lindenmayer *et al.* 2007, Palmer *et al.* 2010). However, many stream and river restoration projects fail to accomplish their objectives and inadequate outcomes are reported from many efforts (Bond and Downes 2003, Palmer *et al.* 2005, Roni *et al.* 2008). Therefore, the aim of this thesis was to investigate the factors that may slow or limit the recovery of restoration efforts in degraded streams.

Concerns surrounding the debate over the broadening definition of what counts as restoration suggest that this confusion may contribute to the inadequacy of restoration efforts occurring globally (Palmer and Filoso 2009, Vörösmarty *et al.* 2010). According to Roni *et al.* (2008), the term restoration is often used loosely in ecology, and should refer to the return of an ecosystem to its original pre-disturbed state. Therefore, many activities associated with restoration are more accurately termed rehabilitation. An alternative approach is that restoration activities occur along a continuum over different levels of degradation and the return to a pre-degraded state is just one point along of this continuum (Hobbs and Norton 1996). Other authors describe ecological restoration as the return of a degraded ecosystem to a more “healthy” state. They regard returning an ecosystem to a pre-degraded state, given the current state of science, as unrealistic regardless of whether it is a goal or not (Palmer and Filoso 2009). Controversy around terminology, described by Hobbs and Norton (1996) as “endless quibbling”, should not diminish the importance of restoration and importantly we need to learn from all attempts and outcomes. The different definitions surrounding these ecological terms can often be confusing, but we mostly just need to get on with the job. Originally, the term restoration was a profession devoted to the preservation of something for the future, which is an important focal point for conservation, and seems an appropriate umbrella term to use here. Therefore, I defined restoration as the return of a degraded ecosystem to a more healthy state.

The numerous stream restoration projects undertaken in recent times have seen stream restoration described as a profitable business (Bernhardt *et al.* 2005, Jähnig *et al.* 2011). However, many of these projects designed to restore rivers are currently being conducted with minimal scientific context and associated unrealistic assumptions (Wohl *et al.* 2005, Jähnig *et al.* 2011). The assumptions that habitat heterogeneity promotes species diversity (Harper and Everard 1998, Palmer *et al.* 2010), that species will naturally return once restoration activities have been completed, and that structural improvements will promote water quality are all linked to a few successful projects, whereas expectations are grand and often unrealistic (Palmer *et al.* 1997, Palmer *et al.* 2010).

Many restoration projects focus solely on habitat enhancements and modifications, and therefore, restoration of in-stream habitat is one of the more common river restoration practices (Purcell *et al.* 2002, Bernhardt *et al.* 2007). Typically, the goal of in-stream habitat restoration is to increase the diversity of aquatic organisms through enhanced substrate and flow heterogeneity or increased food availability (Lepori *et al.* 2005, Roni *et al.* 2006). In streams with largely homogenised environments, habitat restoration is most commonly focused on habitat additions, such as boulder or wood additions and channel reconfiguration at the reach-scale (Miller *et al.* 2010).

Habitat plays an important ecological role in streams but many other factors also influence stream biodiversity, such as high pollutant loads (sediment, nutrients, heavy metals), degraded hydrological regimes, and lack of a colonist species pool (Palmer *et al.* 2010). For example, habitat modifications may not have the chance to influence diversity when there are other more limiting factors (Palmer *et al.* 2010). Habitat heterogeneity may have more important influences when coupled with other features that change diversity, such as disturbance, food resources, and regional species pools (Lake 2000, Ward and Tockner 2001, Hoffmann and Resh 2003, Wohl *et al.* 2005). However, other factors such as riparian vegetation and landscape structure may be just as or more important than habitat diversity (Urban *et al.* 2006). Therefore, diversity and composition of stream communities depend strongly on factors at multiple scales within catchments, which suggests more must be done than just enhancing local habitat complexity and that restoration efforts must target the most limiting factor before stream biodiversity can be improved (Townsend *et al.* 2003, Townsend *et al.* 2004, Palmer *et al.* 2010, Greenwood *et al.* 2012).

There are currently many methods and techniques developed to restore freshwater habitats with foci ranging from large-scale attempts to remediate natural processes right through to local

habitat alterations aimed at enhancing habitat for specific species (Roni and Quinn 2001). Complete removal of livestock and fencing have proven positive influences on bank stability and channel features, including substrate embeddedness, depth/width ratios, and riparian vegetation (Roni *et al.* 2008, Herbst *et al.* 2012). The majority of successful restoration projects suggest in-stream enhancement projects, which when properly implemented *can* potentially produce dramatic improvements in habitat and biodiversity (Roni *et al.* 2008). When enhancements are coupled with solutions to larger scale issues, local habitat improvement can lead to the long-term recovery of hydrology and water quality, sediment transport and riparian conditions. Therefore, the most successful projects create large changes that reflect natural processes (Roni and Quinn 2001, Roni *et al.* 2008). The challenge is to improve knowledge so that barriers preventing these sorts of outcomes can be overcome.

In this thesis, a Canterbury high country stream impacted by agriculture was observed in order to investigate the factors that may slow or limit the recovery of restoration efforts. This focus stream, Riversdale Stream, is surrounded by high quality aquatic habitat, therefore, I predicted surrounding streams should contain a diverse assemblage of aquatic insects which, would provide a connection to a species pool for recolonisation. Assuming an intact connection, in-stream habitat may limit benthic communities. I hypothesised that degraded, in-stream habitat and lack of oviposition would limit recolonisation, and that due to the hysteresis effect of accumulated sediment, additional restoration techniques would need to be improved or developed to bring about restoration.

CASE STUDY

Riversdale Stream, located on Riversdale flat, is in the Cass region of the upper Waimakariri River valley, Canterbury high country, New Zealand (Figure 1.). Riversdale Stream is spring-fed with high quality ground water that is dependent on catchment rainfall and runs through both conservation land (Department of Conservation) and private land managed by Mount White Station. The surrounding area combines native beech forest from Arthur's Pass National Park, farmed grassland, and the large braided Waimakariri River (Figure 1.). The grassland is vegetated by modified tussock (native and introduced grasses) and sporadic short woody shrubs, such as *Discaria* (Burrows 1977). The national park grassland has been highly modified since the 1850's when it was mostly cleared for low intensity agriculture, such as sheep farming (Burrows and Lord 1993). Initially fire was used to clear unwanted

vegetation (Burrows 1977), but burning of this area can be dated back prior to European settlement, where Polynesian fires resulted in a profound modification to the landscape (Burrows 1977). Prior to clearance, the Cass basin was largely covered in beech and conifer-broadleaved forests (McGlone 1989, Burrows and Lord 1993), which are now largely restricted to Arthur's Pass National Park and other conservation land. Today this general area consists of, mostly low intensity agriculture, very little urban areas, and conservation land.

The immediate land-use surrounding Riversdale Stream is pasture seasonally grazed by cattle (April-August) and sheep (year round). The stream is unprotected from grazing animals because there is no immediate fencing or riparian buffer. The stream currently appears to be in a highly degraded state illustrating the degradation commonly associated with agriculture and caused via impacts from livestock grazing (Herbst *et al.* 2012). The banks are heavily eroded and slumped and there is also evidence of slouching of the surrounding land, reduction of vegetation cover causing breaks, bare soil, increased water temperature, reduced dissolved oxygen, nutrient enrichment, and frequent filamentous algal growth (Figure 2 and 3.).

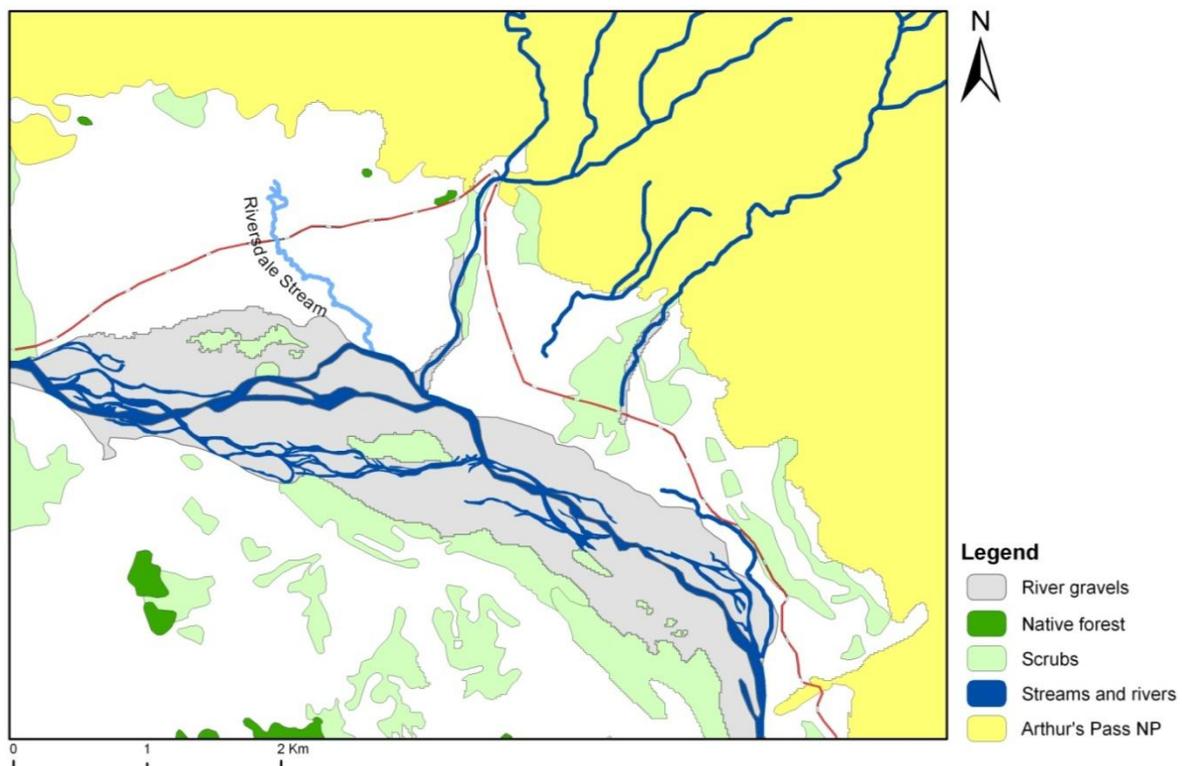


Figure 1. Riversdale Stream in the Canterbury high country of New Zealand, surrounded by high quality habitat of Arthur's Pass National Park (NP) beech forest, other native forest patches, woody scrubs such as matagouri, the Waimakariri River, and other streams beginning in the beech forest.

Heavy sedimentation has occurred along the majority of the stream due to this bank erosion, with deposits up to 20 cm deep (Figure 2 and 4.). These excessive fine sediment loads have impacted benthic habitat and have likely altered invertebrate communities, particularly through the reduction of stream insect populations sensitive to sediment (Rabeni *et al.* 2005). The deposited sediment has also possibly caused this stream to be moved into a state of hysteresis, whereby the sediment will require specific restoration efforts to remove it before returning to a less degraded state is possible (Suding and Hobbs 2009).



Figure 2. The banks of Riversdale Stream are heavily eroded and slumped allowing sedimentation. Additionally due to over-widening and low bank angles at certain areas of the stream, when high flows wash through they often flood the surrounding land, rather than flushing the stream of deposited sediments. Photos b, and c, by Angus McIntosh.

Many restoration projects do little in the way of encouraging sediment removal, but focus on reducing further inputs (Lowrance *et al.* 1984, Dillaha *et al.* 1989, Craig *et al.* 2008) assuming streams possess the natural capacity necessary to remove the sediment build-up (Greenwood *et al.* 2012). However, in many situations, as seen in Canterbury Plains, restoration efforts such as riparian management are often not sufficient at removing sediment build-up. In these cases, in-stream

damage may be beyond the influence of these restoration techniques and removing the potential threats may not result in fine sediment reductions (Greenwood *et al.* 2012).



Figure 3. Extensive macrophyte growth (dark green) at Riversdale Stream slows velocity and allows long strands of filamentous algal growth (lighter green) as shown.

In addition, the lack of riparian vegetation surrounding Riversdale Stream means there is very little shade along the stream channel allowing extensive macrophyte growth (Figure 5.). Excessive growth generally restricts flow and creates areas of pooling (Bunn *et al.* 1998). These stands of macrophytes then further trap sediments and can advance adverse changes in-stream morphology and hydrology in areas already degraded via over-widening of the bed caused by cattle trampling (Bunn *et al.* 1998). Although increased nutrient and sediment inputs from livestock are also likely to stimulate in-stream macrophyte production, light availability is considered the primary factor controlling macrophyte distribution and abundance (Bunn *et al.* 1998). Therefore, excessive macrophyte growth and associated sediment trapping may be controlled or reduced via shading produced by riparian vegetation.



Figure 4. a) Underwater view of in-stream sediment at Riversdale Stream, b) top view of stream sediment, showing 100% coverage over the majority of the stream and very little in-stream habitat complexity.



Figure 5. Excessive macrophyte growth at Riversdale Stream restricts flow and creates areas of pooling allowing further sediment trapping, this can also advance changes in stream morphology and hydrology to form widened channels and further reduce velocity. a-d display channel clogging and e, underwater view of macrophyte channel clogging.

Stream from regime

Rainfall in the area is principally a-seasonal and averages 1300 mm per annum (Burrows 1977). Riversdale Stream's flow is dependent on this rainfall, and often during periods of low precipitation, this stream may have low discharge, drying into pools or drying completely (Figure 6.). Flow intermittency may also be exacerbated by reductions in shade and interstitial clogging. Interstices clogged by sediment remove invertebrate refugia and reduce water exchange with ground water that drives the springs. Moreover, during drought and low-flows, habitat availability and refugia are vital for population persistence, and the capacity for individuals to survive depends upon their ability to use available habitat effectively (Lake 2000, Arthington *et al.* 2005, Boulton and Hancock 2006, Bunn *et al.* 2006, Storey and Quinn 2011).

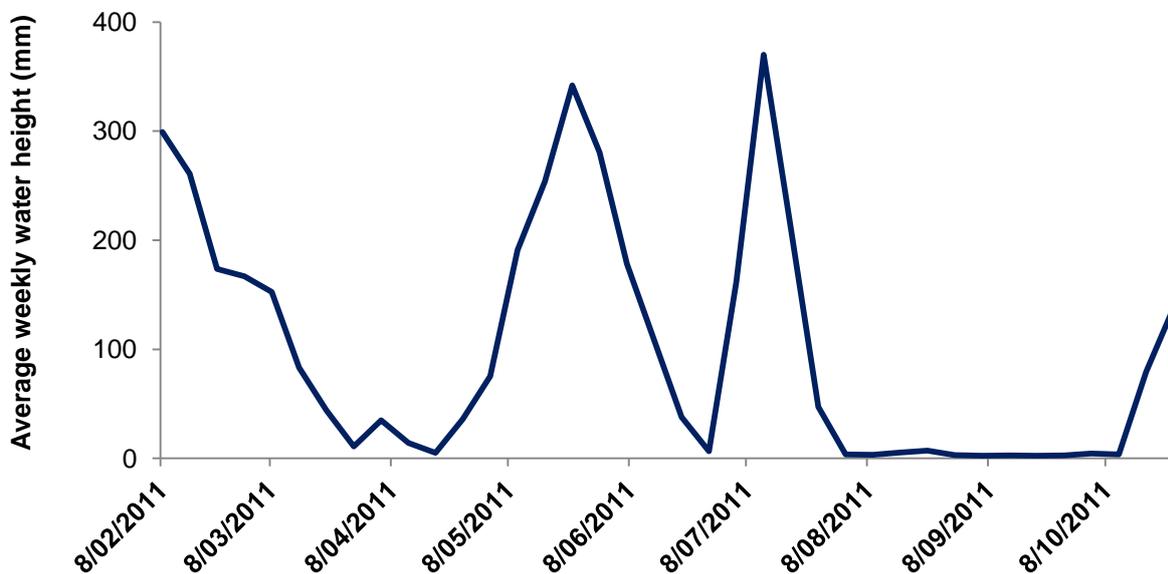


Figure 6. Average weekly water height from a stage height logger at Riversdale Stream from February-October 2011, shows various changes of flow that coincide with rainfall events.

Aquatic refugia take a variety of forms and some species are better suited to withstand these habitats under disturbance. For example native galaxiids can withstand habitat drying better in comparison to introduced trout (McIntosh 2000, McIntosh *et al.* 2010). Therefore, intermittent streams such as Riversdale Stream can be important refugia from introduced species as extreme flow disturbances can reduce or remove the effect of introduced trout (McIntosh 2000, Davey *et al.* 2006, McIntosh *et al.* 2010). Galaxiids can persist in intermittent habitat by using refugia and during extreme low-flow events by burrowing into the stream substrates (Davey *et al.* 2006, Davey and Kelly 2007). However, during land-use change habitat degradation, such as sedimentation decrease the

availability of suitable refugia (Lake 2003, Bond and Lake 2005, Bond *et al.* 2008). Therefore, Riversdale Stream is currently not likely to provide an introduced species free habitat for native fish (Figure 7.). Nonetheless, Riversdale Stream may be an important location for the protection of New Zealand's native fish, which hinges on the creation of drought refugia during restoration.



Figure 7. Poor habitat available at Riversdale Stream during periods of very low flow events restricts survival of stream biota especially for native fish.

Species pool and habitat fragmentation

As mentioned earlier, one common underlying assumption of stream restoration regards the automatic return of biodiversity post-habitat modification (Violin *et al.* 2011). This assumption is based on habitat heterogeneity and species richness being positively correlated and has been termed the field of dreams hypothesis, where "if you build it, they will come" (Palmer *et al.* 1997, Brown 2003, Helfield *et al.* 2007, Kail *et al.* 2007). One possible explanation for the failure of this assumption is lack of connectivity to the regional species pool, implying that individuals will have to be present locally to be able to colonise a restoration site (Miller *et al.* 2010). However, a limited species pool may be

present in many agricultural streams and therefore the presence of this connection may warrant restoration priority. Beyond the agricultural land-use directly surrounding Riversdale Stream is Arthur's Pass National Park, Craigieburn Forest Park, and the Waimakariri River, which may allow for the dispersal of a variety of aquatic insect species in the regional pool. This connection may be disproportionately important for the potential recovery of aquatic insects during any restoration efforts at Riversdale Stream. In addition, previous studies focusing on the terrestrial stages of aquatic insects from this area, such as flight patterns, distribution, and dispersal, give evidence of a diverse assemblage making up the regional species pool (Winterbourn and Crowe 2001, Winterbourn 2007, Winterbourn *et al.* 2007). Thus, an important step in determining the potential for restoration at Riversdale Stream is to establish whether there is in fact a connection to a diverse species pool.

The role of land-use change in limiting colonisation and dispersal to date has been poorly studied (Briers *et al.* 2002). However, the conclusions made in a review by Didham *et al.* (2012) suggest that longer distance dispersal is more common than originally thought. However, dispersal across different land-use types is less common and community compositions among different land-uses differ, suggesting this may limit colonisation despite species presence (Briers *et al.* 2002, Winterbourn *et al.* 2007, Smith *et al.* 2009, Didham *et al.* 2012). In addition, the likelihood of colonisation to a stream is also dependent on the habitat available within the stream. For example, lack of appropriate oviposition habitat can limit whether a stream is colonised (Reich and Downes 2004), and species presence also may be dependent on emergence and flight times of adult aquatic insects and whether this corresponds with base flow periods of intermittent streams (Winterbourn and Crowe 2001).

Palmer *et al.* (2010), in a comprehensive meta-analysis of 78 independent restoration projects, found two of these resulted in sufficient increases in invertebrate diversity for the author to conclude the project was a success. Despite the broad range of restoration techniques, several common factors appear to limit the success of projects. Water quality and quantity, erosion and sedimentation, fragmentation, and loss of connectivity to species pools prevent many projects from achieving their full potential (Wood and Armitage 1997, Winterbourn *et al.* 2007, Roni *et al.* 2008, Palmer *et al.* 2010, Didham *et al.* 2012). Each of these limiting factors that result in in-stream restoration failure, reflects a lack of understanding of surrounding broader scale issues, reinforcing the need to consider and

prioritise larger scale problems (Beechie and Bolton 1999, Roni *et al.* 2002, Wohl *et al.* 2005, Beechie *et al.* 2008).

Restoration priorities

Beechie *et al.* (2008) suggest logical and analytical approaches to prioritising stream restoration, including single and multiple species approaches to restoration as well as a method based on cost efficiency and a scoring system to weigh up pros and cons of different potential projects (Beechie and Bolton 1999, Bohn and Kershner 2002, Roni *et al.* 2002, Cipollini *et al.* 2005). However, prioritising streams based on their probability of success, i.e. maintaining and improving good habitat, (Roni *et al.* 2002, Roni *et al.* 2008) and selecting streams with close proximity to colonisation source (Huxel and Hastings 1999) ensures restoration efforts should result in rapid recovery (Beechie *et al.* 2008). However ultimately, heavily degraded or polluted streams in headwaters are also likely to contribute greater pollution to downstream habitats and larger rivers (Dodds and Oakes 2008). This therefore makes these heavily degraded headwater streams, such as Riversdale Stream, an additional priority for restoration activities, although restoration goals should be more conservative and realistic.

Furthermore, a stream, such as Riversdale, that may be linked to a diverse regional species pool and a water source that is unlikely to be affected by poor water quality is expected to benefit from current restoration techniques, such as fencing, riparian vegetation plantings, and habitat addition (i.e. wood and boulders). Riparian vegetation may recover when grazing pressure is removed or reduced, however, channel structure and in-stream recovery, particularly from heavy sediment deposits, may take longer with outcomes depending on immediate land-use, sediment sources, and the timing of sediment-flushing flows (Herbst *et al.* 2012). In addition, some streams, possibly Riversdale, may have become so degraded that they are unable to recover naturally after the degradation pressure i.e. livestock, has been removed (Laycock 1991). Therefore, restoration of this stream will likely require attention to the issue of bank erosion, channel widening, deposited sediment and further testing of assumptions.

In summary, Riversdale Stream, a stream surrounded by high quality habitat and possibly a diverse assemblage of aquatic insects, should follow the field of dreams hypothesis “if we build it they will come” making it an ideal study site for testing restoration techniques and theories. I hypothesised that degraded in-stream habitat and lack of oviposition would limit recolonisation and that, due to the

hysteresis effect of sedimentation, the removal of fine sediment from the stream would result in the recovery of sensitive invertebrates if sedimentation was the most limiting factor restricting populations. In addition, I predicted that the combination of mechanical and hydrological movement of sediment would be most effective, and therefore proving the best quality habitat for sensitive insect colonisation.

THESIS STRUCTURE

The main aim of my thesis is to understand what can limit invertebrate recovery during stream restoration using Riversdale Stream as a case study. This thesis is written as standalone papers that will be submitted for publication, and consequently some of the chapters share similar material. However, to minimise repetition I have referenced previous chapters in some situations. In Chapter One, I have briefly introduced the state of stream and river restoration and outlined the need for restoration at Riversdale Stream. In Chapter Two, I tested the assumption of the field of dreams hypothesis and relate this to the condition of Riversdale Stream. In Chapter Three, I investigated what is limiting the recovery of invertebrate communities at Riversdale Stream and trialled methods to aid recovery. Finally, in Chapter Four, I combined my results and conclusions to form management recommendations, not only for Riversdale Stream, but also for other streams that may be in similar condition and candidates for stream restoration in general.

CHAPTER TWO

Factors limiting colonisation and recovery of sensitive stream insects during stream restoration



View of the Waimakariri River on the Mount White Road Bride

INTRODUCTION

With growing awareness of degradation of natural habitat comes our increased desire and efforts to restore those environments. Restoration describes an umbrella of specific management techniques (Bernhardt *et al.* 2007, Brederveld *et al.* 2011) designed to speed up or bypass intermediate succession phases to produce a climax or desired community (Aerts and Honnay 2011). To avoid getting too bogged down in the restoration/rehabilitation terminology discussion, “restoration” is used here in the broadest context covering rehabilitation, recreation, remediation, re-vegetation, recolonisation, reintroduction, recovery, and reconstruction (Hobbs and Cramer 2008). The ecosystem approach to restoration aims to restore abiotic conditions to facilitate passive recolonisation of species (Aerts and Honnay 2011). Using this approach, connections, barriers, and fragmentation of habitats can limit community response or recovery (Aerts and Honnay 2011), so it is important to better understand their influence.

In terms of stream and rivers, restoration techniques often include but are not limited to: replanting and fencing riparian zones (livestock exclusion), removal or enhancement of man-made structures near water bodies (Rohde *et al.* 2005), bank stabilisation, addition or enhancements of in-stream habitat (Kail *et al.* 2012), and full scale redesign of stream channels (Gurnell *et al.* 2006). River restoration management has even been called a profitable business (Bernhardt *et al.* 2005), where projects are often conducted with minimal scientific input (Palmer *et al.* 2005, Wohl *et al.* 2005). However, in most cases, the desired outcomes of biodiversity improvements are not met (Larson *et al.* 2001, Rohde *et al.* 2005, Bernhardt and Palmer 2007, Spänhoff and Arle 2007, Jähnig *et al.* 2009, Palmer *et al.* 2010). Therefore, restoration practices, techniques, and associated assumptions require thorough scientific testing to determine whether efforts will lead to desired ecological outcomes (Hobbs 2005).

One common underlying assumption is that restoration of stream morphology to a pre-degraded state will lead to the recovery of aquatic organisms (Violin *et al.* 2011). This assumption is based on research demonstrating habitat heterogeneity and species richness are positively correlated (Brown 2003, Helfield *et al.* 2007, Kail *et al.* 2007). This approach has been termed the “field of dreams hypothesis”, after the 1989 movie, whose catch-phrase was “If you build it, they [originally, he] will come”. While this movie considers a baseball field, the field of restoration ecology is also filled with dreams (Palmer *et al.* 1997). Following this assumption, many stream restoration projects are

designed to increase habitat heterogeneity as their primary mechanism of biodiversity restoration, despite experimental evidence suggesting substrate variability does not always result in species recovery (Palmer *et al.* 1997, Larson *et al.* 2001, Brooks *et al.* 2002, Blakely and Harding 2005, Suren and McMurtrie 2005, Spänhoff *et al.* 2006, Bernhardt and Palmer 2007, Sudduth *et al.* 2011). This is not to say that improving habitat structure is not important; effective restoration projects should recapture the habitat structure and particularly the complexity of reference or pre-degraded conditions (Suding *et al.* 2004, Palmer *et al.* 2005). However, a question begging to be asked 'is habitat restoration in streams and rivers alone adequate?' This question is particularly relevant because many freshwater bodies are located in highly degraded, fragmented, and often disjointed landscapes (Palmer *et al.* 1997). With this in mind, determining whether connectivity conservation (such as provision of corridors or species translocation) should take priority over habitat improvement is particularly important (Jansson *et al.* 2000, Milner *et al.* 2000, Renöfält *et al.* 2005, Rohde *et al.* 2005, Lake *et al.* 2007).

Miller *et al.* (2010) found high variability in post-restoration outcomes of benthic communities and suggested restoration failure was associated with lack of connectivity to the regional species pool. A species pool can connect aquatic insect populations during both larva (drift and crawling) and adult (flight) life-stages (Verberk *et al.* 2008). Recolonisation via the larval life-stage requires an intact species pool upstream or nearby, which is often not available in degraded catchments (Smith *et al.* 2009). Therefore, adults are seen as the primary recolonisation stage in the recovery phase of most aquatic insect populations (Williams and Hynes 1976). However, colonisation limitation can take two forms, firstly, colonisation can be source-limited, whereby a species is absent from the species pool or has limited dispersal capabilities. Secondly, a species may arrive at a site but be limited by a specific habitat requirement that restricts oviposition or larval survival in that habitat.

Consequently, adult aquatic insect dispersal is dependent on physical adaptations as well as behaviour, landscape structure and arrangement, and in-stream habitat (Sode and Wieberg-Larsen 1993, Downes and Keough 1998, Malmqvist 2000, Bohonak and Jenkins 2003). Fragmentation and degradation of habitats in riparian landscapes can disrupt dispersal strategies, reducing colonisation likelihood and survival during dispersal (Epps *et al.* 2007). Even within insect groups, there is considerable variation in flying ability, resulting in differences in dispersal (Lancaster and Briers 2008).

Although abundance of individuals in the local and regional species pools are important factors predicting colonisation success (Brederveld *et al.* 2011), organisms with complex life cycles that form open populations, such as aquatic insects, are also strongly influenced by dispersal and stochastic processes driving recruitment but density dependence may also be important (Palmer *et al.* 1996, Hildrew *et al.* 2004). Bunn and Hughes (1997), for example, found that local aquatic insect populations could be the result of a few successful reproducing females. Species with higher abundances of adults are probably more likely to colonise restored sections of stream, although successful in-stream populations can arise with the dispersal and oviposition of a few individuals, however colonisation may also be limited by egg-laying opportunities (Hughes *et al.* 2009).

Egg-laying (oviposition) may be a key bottleneck in the life cycle and availability of oviposition habitat, the habitat in which aquatic insects lay their eggs, which may be critical (Reich and Downes 2004, Smith *et al.* 2009). Oviposition preferences and behaviour vary and are often dependent on factors such as water depth and velocity, but frequently involve overhanging riparian vegetation or emergent stream substrate or logs (Collier *et al.* 1995, Peckarsky *et al.* 2000, Reich and Downes 2004). Hydrobiosid caddisflies (Trichoptera), for example, lay aggregated egg masses on the underside of emergent rocks with individual species expressing different water velocity preferences, which can influence the location of early instar larvae (Reich and Downes 2004, Lancaster *et al.* 2010). This demonstrates the importance of diverse in-stream habitat for oviposition of different species. Given dispersing adults are also vital for recolonisation, the lack of either adults or oviposition habitat could potentially limit the recovery of a degraded stream post-restoration (Bunn and Hughes 1997, Purcell *et al.* 2002, Bond and Lake 2003).

Degradation of terrestrial and aquatic environments can inhibit all life-stages of aquatic insects (Smith *et al.* 2009). Aquatic invertebrate species vary in their habitat requirements and tolerances to pollution, and therefore can only occupy a certain subset of streams in a degradation continuum (Collier *et al.* 1995). Land-use change to agriculture and intensification can have corresponding negative impacts on benthic communities which reflect the terrestrial environment (Collier *et al.* 1997). Therefore, sensitive taxa, such as Ephemeroptera, Plecoptera and Trichoptera (EPT), are usually not abundant in highly degraded conditions and make appropriate indicators to detect habitat improvement during restoration to determine success (Gamboa *et al.* 2008). However, as previously discussed above, we cannot assume species return because of a number of influencing factors.

Bond and Lake (2003) concluded their review on restoration outcomes by stressing the importance of the interaction between habitat patch structure, dispersal, and colonisation and how these drive responses to restoration. Aquatic insect populations are dependent on colonisation events and distance from potential source populations (Bond and Lake 2003). Therefore, a community that remains connected to an intact regional species pool is expected to show a significant positive response to habitat restoration even in highly degraded sites (Miller *et al.* 2010). I aim to determine the factors limiting benthic EPT communities in a degraded stream surrounded by high quality habitat. Therefore, if a species is found in the species pool but not at the desired restoration location, it may be concluded that there is a habitat- and not a colonisation-limitation (Brederveld *et al.* 2011). I examined both of these processes in this study. Firstly, I identified the species that comprised the adult regional pool from a Canterbury high country stream impacted by agriculture to determine if the availability of colonists limited the return of larvae to the benthic community. This stream, Riversdale Stream, is surrounded by high quality habitat in the adjacent Arthur's Pass National Park and therefore, I predicted surrounding streams should contain a diverse assemblage of aquatic insects. Secondly, I evaluated areas of in-stream habitat that may also limit benthic communities such as quality of the benthic habitat, and availability of oviposition habitat. Here I hypothesised that degraded in-stream habitat and lack of oviposition would limit recolonisation using hydrobiosid caddisflies as an example.

METHODS

Sites and in-stream habitat quality

Riversdale Stream sites were selected at 100 m intervals upstream and downstream of the Mount White Road Bridge adjacent to Arthur's Pass National Park in the South Island high country of New Zealand (Figure 1.). Two sites (Site One and Two) were located upstream and the remaining sites (Site Three, Four and Five) were down stream of the bridge. In addition, three surrounding streams of different habitat quality and type were selected as reference sites. A forested stream, Reference One (Peacock Stream), a scrubland stream, Reference Two (Lower Farm Stream) and a grassland spring stream, Reference Three (Waimakairi Spring), all contained one sampling site (Figure 1.). Local habitat variables were measured to evaluate the habitat condition of each site using a modified habitat assessment protocol developed for New Zealand streams (Harding *et al.* 2009). This habitat survey resulted in a score that could place the sampling site within four categories of habitat condition,

ranging from excellent to poor. This involved visually assessing the quality of the surrounding land-use, riparian width, type and continuity, bank stability, degree of pugging and bank slumping, in-stream habitat and cover, shadiness, embeddedness, deposition, fine sediment cover, macrophyte, algae and organic matter type and presence. In addition, I measured substrate heterogeneity using the Wolman pebble count method with 25 particles, fine sediment depth, and flow diversity (using a Marsh McBirney flow meter). Chemical water quality characteristics were also measured with a handheld YSI multi-parameter system and included specific conductivity (Ys/cm at 25 °C), temperature (°C), dissolved oxygen (mg/l) and pH.

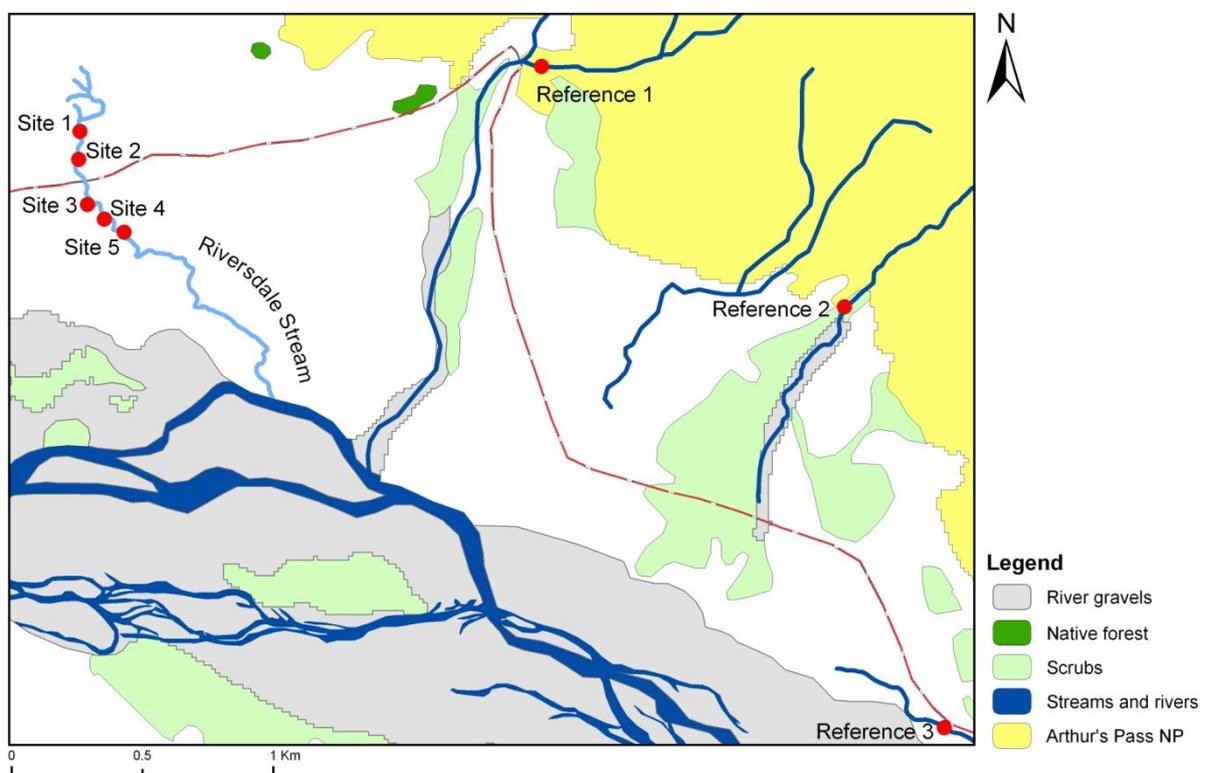


Figure 1. Site map, displaying Riversdale Stream's five sites, and surrounding high quality habitat that incorporates the three reference sites. Reference One, Peacock Stream a beech forested stream site in Arthur's Pass National Park (NP), Reference Two, Lower Farm Stream a scrubland site, exiting the national park and Reference Three, Waimakariri Spring, a grassland Spring-fed stream. In addition, this map also shows the Waimakariri River bed and associated high quality habitat.

Larvae and adult population survey

Surber samples (0.05 m², 500 µm mesh) were collected for the five Riversdale and three reference sites to evaluate larvae communities following general sampling guidelines (Harding *et al.* 2009). Samples were collected in January 2011, corresponding with adult capture times, and stored in

ethanol until identified in the Laboratory. Benthic macroinvertebrates were identified to genus or species using keys in Winterbourn *et al.* (2006) at a 10-30 x magnification under a dissection microscope.

To survey the adult aquatic insect community and their dispersal, malaise traps (1.9 m high, with 36.8 m³ of trapping area) were set up at 0 m, 10 m, 20 m from the stream edge. Traps were set in January through March 2011 and collected weekly. In addition, two malaise traps were set out 100 m from the stream edge at Riversdale Stream towards Arthur's Pass National Park and towards the Waimakariri River bed. Malaise traps were ideally designed to gauge dispersal to Riversdale Stream. However, this sampling method failed to capture substantial numbers of aquatic insects at any site and therefore these data were not used for analysis.

In addition, light traps were also used to survey the adult community and were set up between Sites One and Two at Riversdale Stream, an area considered to have highly impacted in-stream habitat and again between Site Three and Four an area with less or low impact, and the three reference sites. Light traps were constructed from garden lights adapted with UV-LEDs (black light, 395–400 nm, a wave length known to attract flying aquatic insects; Collier *et al.* 1997). The garden light was secured in the centre of a white plastic tray (24 x 35 x 7 cm) half filled with water and a few drops of detergent to break water surface tension (Smith *et al.* 2002). Traps were placed in open areas where possible, to avoid vegetation interference, as light trapping effectiveness is reduced with increasing vegetation density (Collier *et al.* 1997). Light trapping was restricted to nights where weather conditions were predicted to be above 9 °C with light winds (Collier and Smith 1997) between January and March 2011 (six trapping nights) and 2012 (three trapping nights). Light traps were most successful in attracting and capturing Trichoptera, so analyses were confined to this order and adult Trichoptera were identified to family level (Neboiss 1986) using 10-30 x magnification under a dissection microscope.

Oviposition experiment

To determine if oviposition habitat limited recruitment of sensitive species recovery at Riversdale Stream, ten boulders were added per site of each to the five Riversdale Stream sites. A site consisted of a 20 m stretch of stream divided into two sections, with the upstream section nominated to be the control section and downstream the manipulation section. Hydrobiosidae caddisflies are known to use

emergent substrate as oviposition habitat (Reich and Downes 2003). Adult female caddisflies show strong oviposition patterns whereby they almost exclusively lay a single, jelly-covered egg mass on the underside of emergent boulders (Reich and Downes 2003, Verberk *et al.* 2008). Boulders were added approximately 1 m apart, at a random distance from the true right side bank. Boulders were sourced from a nearby section of the Waimakariri Riverbed and sized, depending on the depth of each site, to ensure emergence at base flow. Boulders were checked for egg masses every fortnight in February and March 2011 and again in January and February 2012, when flow conditions allowed. Egg masses are easily detected via searching the boulder surface and fortnightly sampling was selected to ensure a single egg mass was likely to have hatched before a subsequent count, and not counted twice (Blakely *et al.* 2006). In addition, ten of the largest substrates were also checked for egg masses in each control section to compare natural and added habitat egg mass abundance.

Statistical analysis

I assessed variation in community composition using an ordination of species abundance for (a) the larval, Surber samples (genera diversity, samples as replicates), (b) adult, light trap samples (family diversity, traps as replicates), and (c) combined larvae and adult communities (family diversity). The abundance of each species (square-root transformed to downweight highly abundant species) was used to create a Bray Curtis dissimilarity matrix which was subjected to ordination using non-metric multidimensional scaling (NMDS) with 20 random starts in both two and three dimensions using PRIMER (Clarke and Gorley 2000). This produced measures that represented differences in community composition across samples along the ordination axes. I retained two dimensions in the final model to simplify analysis, as adding the third dimension resulted in a minor decrease in stress.

To further, investigate differences in adult abundance at family level, log-transformed abundance of the five most common families was compared at different levels of impact (high impact, low impact, and reference), initially using a MANOVA to control for the multiple non-independent response variables and then multiple univariate ANOVA's were completed with contrasts using R, V.2.15.0 (Crawley 2007). A priori contrasts were used to compare high impact against low impact (Riversdale sites), reference (all three streams) against Riversdale (both impact levels), and then reference sites with high and low impacted sites separately. These two impact levels were chosen because of the clear environmental gradient at Riversdale Stream in comparison to the other streams.

This analysis was then repeated using two-way ANOVA with life-stage and impact level as factors, testing for differences between adults and larvae life-stages over the two impact levels.

The oviposition experiment was set up as a split-plot analysis, however due to the high impact area attracting almost no eggs violating the assumptions of normality, the highly impacted sites were not tested statistically. However, the drastic and obvious differences in the abundance of egg masses between high and low impact sites made testing unnecessary (Figure 6.). Instead, only the second level of the split-plot analysis was tested as a nested design, where substrate type (added boulders or natural substrate) was nested within the low impact sites with added boulders or natural substrates used as replicates.

RESULTS

Habitat quality

Habitat score calculations revealed the eight sites spanned the four habitat quality categories (Figure 2.). Three low scoring sites (Riversdale Stream Sites One, Two, and Five) fell into the poor habitat quality (severe degradation), while the remaining two Riversdale sites (Site Three and Four) were classed as marginal (moderate degradation). The most striking difference between these two groups of sites was the quantity of fine sediment (<2 mm) dominating the substrate. The poor habitat quality group had sediment depths ranging from 60-160 mm of fines and consisted of 95-100 % sediment cover. The moderate habitat quality group ranged in sediment depth from 0-60 mm in fines and consisted of 20-30 % sediment cover. These groups are referred to as the high- and low-impact sites respectively, from here on. Of the three reference sites, Peacock Stream and Lower Farm Stream (referred to as, Reference One and Reference Two, respectively) were judged to have excellent habitat quality. This habitat quality gradient had substantial effect on in-stream invertebrates; as habitat quality increased so did invertebrate diversity (Figure 2.)

Community patterns

In-stream habitat conditions had a strong positive relationship with in-stream invertebrate diversity (Figure 2.), and the community composition NMDS ordination showed distinctly different communities were present (Figure 3). The reference sites had greater in-stream invertebrate diversity (Figure 2) and more rare species, associated with higher axis 1 ordination scores. NMDS ordination (Figure 3, b.)

showed the five sites form a continuum with Reference Three, intermediate between the high and low impact Riversdale sites and the more pristine reference sites. This pattern was also observed with Trichoptera adult communities and overall there was a significant difference in adult community structure between sites (Table 1.)

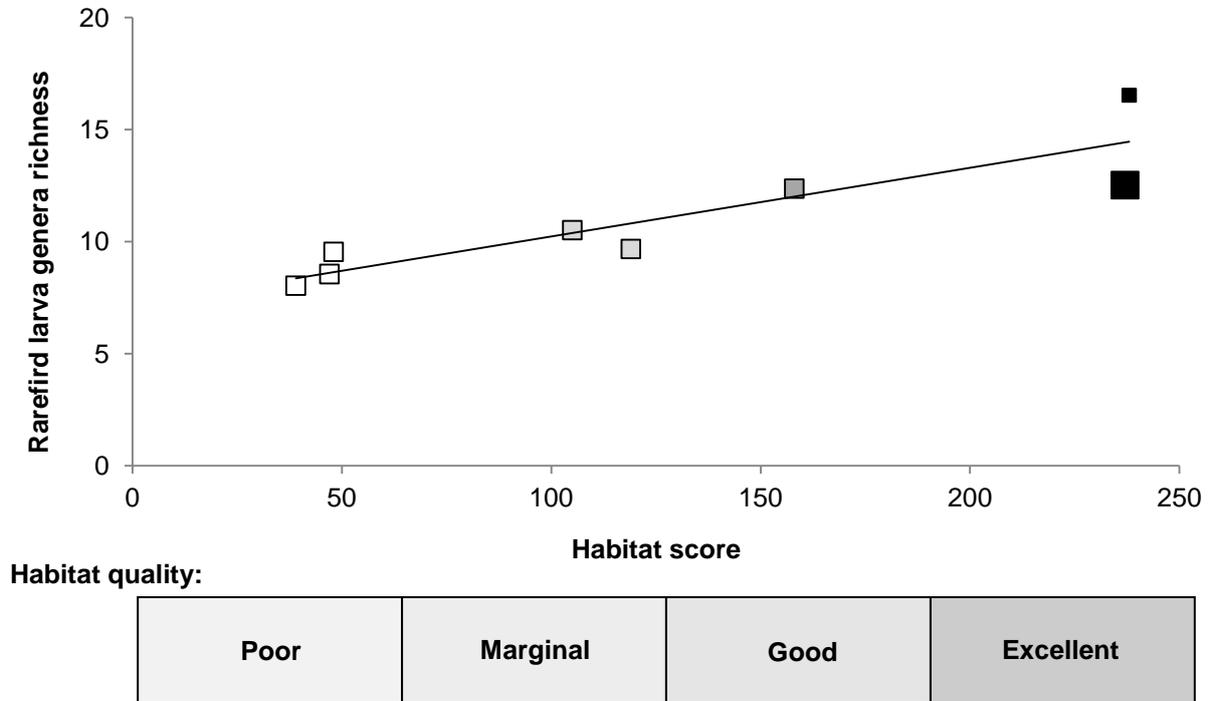


Figure 2. Average rarefied larva genera richness across all sampled sites compared to habitat scores for each site; $P < 0.05$, $R^2 = 0.81$, linear Equation, $y = 0.031x + 7.175$. Habitat scores were calculated using a modified New Zealand standard from Harding *et al.* (2009). Riversdale sites were classed as either poor (three sites) or marginal (two sites) habitat qualities, represented by pale shades of grey. Reference sites were made up of good (Reference Three, Waimakariri Spring) and excellent habitat qualities indicated by dark grey (one site) and black (two sites), respectively. The excellent habitat quality group is made up from two streams and therefore the Reference One, represented by a smaller symbol and Reference Two is represented by the larger symbol.

A detailed previous study on adult stream insect flight in this area, with higher resolution taxonomy (Winterbourn and Crowe 2001), indicated nine Trichoptera families should be found. All nine families were found at the three reference sites, giving the reference sites the highest total adult richness (Figure 4). Waimakariri Spring (or Reference Three) generally had the most abundant catches while the greatest richness was found at the more pristine reference sites (Figure 5.). Seven families were found at the low impact Riversdale Stream site (Philorheithridae and Polycentropodidae absent) and five at the high impact site (Oeconesidae, Philopotamidae, Philorheithridae and

Polycentropodidae absent; Figure 4). Moreover, an intact regional species pool was indicated by adult richness being significantly larger than corresponding larval richness at the family level at all sites (Figure 5.). The most abundant adult Trichoptera families I caught in this study were Hydroptilidae (80 % total catch), Hydrobiosidae (8 %), Conoesucidae (6 %), Hydropsychidae (2 %), and Leptoceridae (2 %), and these families formed the basis of further investigations (Figure 4.).

Overall, there was a significant difference in adult community structure between sites, with MANOVA indicating level of impact (high, low, or reference) significantly affected population abundances of the five most common Trichoptera adult families (Table 1.). Univariate analysis of variance (ANOVA) was used to test the response of families separately, and indicated highly significant differences in adult populations of these families across impact levels (Table 1.). This ultimately indicated that each site had a different adult community structure, verifying the obvious patterns of community structure seen in the NMDS ordination (Figure 3.). However, each family responded to the varying levels of impact slightly differently (Table 1.).

Hydrobiosidae and Leptoceridae adults were more abundant at the less impacted site of Riversdale Stream in comparison to the high impacted site, however Hydrobiosidae abundance at the low impact site was similar to the combined reference site abundance, and this family appeared to be generally less abundant in degraded habitat (Table 1., Figure 4.). Conversely, Leptoceridae are significantly more abundant at the low impact Riversdale site in comparison to the reference sites where abundances were low (Table 1., Figure 4.). Hydroptilidae had high abundances in the grassland sites of Riversdale Stream and Waimakariri Spring (Reference Three), in comparison to the very low abundances in the pristine reference sites (Figure 4.). Therefore, Hydroptilidae were significantly more abundant at Riversdale than reference site, however this was mostly due to the higher abundance in the low impact Riversdale site (Table 1., Figure 4.). Conoesucidae and Hydropsychidae adult abundance differed between Riversdale and reference sites. However the low Conoesucidae abundance at the high impact site contributed mostly to this because the lower impacted site was not statically different to that of the combined reference sites (Table 1., Figure 4.).

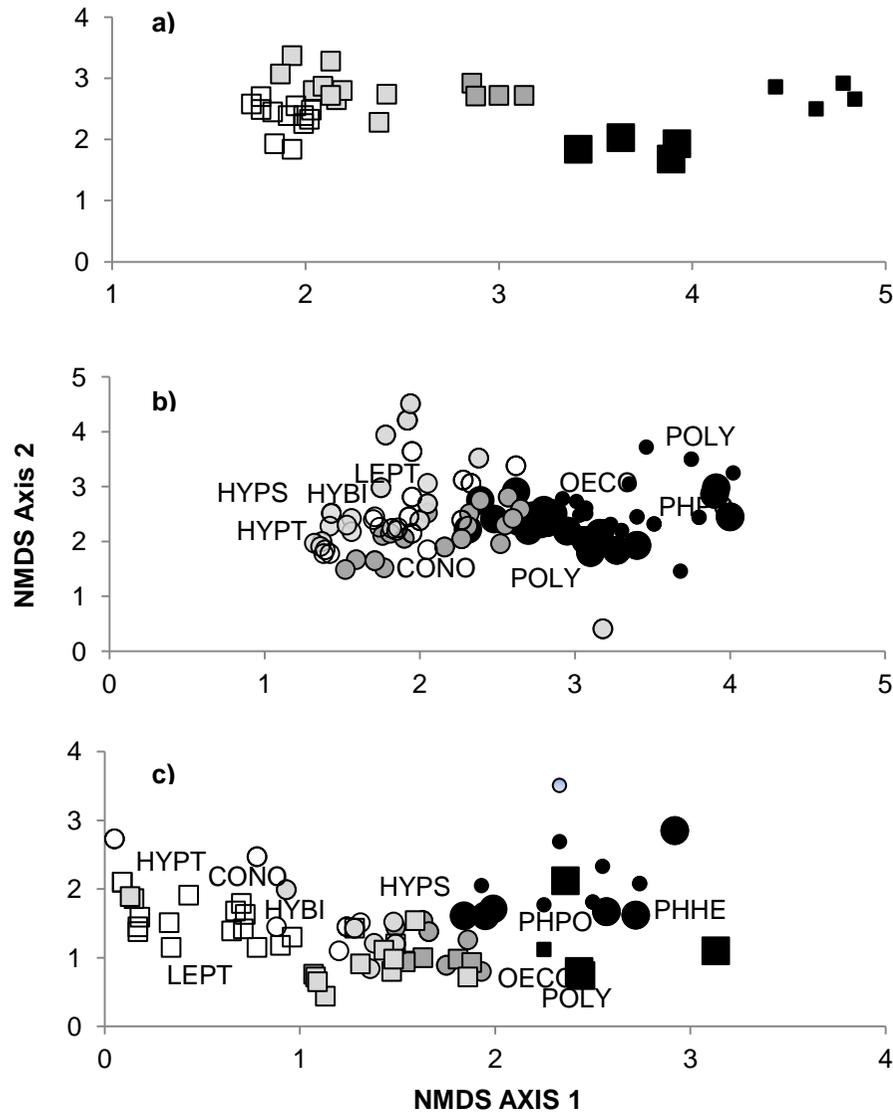


Figure 3. Non-metric multidimensional scaling (NMDS) ordinations of abundance (square-root transformed) for Riversdale and reference stream communities with the four different habitat qualities indicated by shading. a) in-stream invertebrate communities (ordination stress 0.11), b) adult Trichoptera at the family level, influential families represented by letter code (ordination stress 0.12), c) adult and larvae Trichoptera both at family level taxonomy (ordination stress 0.12). High impact sites (three sites) at Riversdale Stream had poor habitat quality (open symbols), and the less impacted Riversdale sites had marginal habitat (pale grey symbols). The reference sites are good (dark grey symbols) or excellent habitat quality (two sites; black symbols). Habitat scores were calculated using a modified New Zealand habitat assessment standard from Harding *et al.* (2009) and grey scaling from open to solid represents the habitat continuum from poor to excellent.

Family Codes

- HYPT, Hydroptilidae
- HYBI, Hydrobiosidae
- HYPS, Hydropsychidae
- CONO, Conoesucidae
- OECO, Oeconesidae
- LEPT, Leptoceridae
- PHPO, Philopotamidae
- PHHE, Philorheithridae
- POLY, Polycentropodidae

Symbol key

	Adults	Larvae
Reference 1	●	■
Reference 2	●	■
Reference 3	●	■
Low impact	○	□
High impact	○	□

Table 1. Using the five most commonly found adult Trichoptera families in this study, a MANOVA and then multiple univariate single factor ANOVA's (abundances were log-transformed) were completed with additional contrasts. Contrast levels were high impact against low impact (Riversdale sites), reference (all three streams) against Riversdale (both impacts) and then reference against high and low impacted sites separately. Sites were determined to be high or low impact using a modified New Zealand habitat assessment standard from Harding *et al.* (2009). *F-ratio for the MANOVA is an approximation calculated by R. The Pillai value for impact is 2.33.

		ANOVA			High impact against low impact Riversdale		Riversdale against reference sites		High impact Riversdale against reference sites		Low impact Riversdale against reference sites	
		df	*F	P	t	P	t	P	t	P	t	P
MANOVA	Impact	4	10.87	<0.01								
	Residuals	40										
Hydroptilidae	Impact	4	14.16	<0.01	-0.94	0.35	-3.07	<0.01	-1.55	0.13	-3.57	<0.01
	Residuals	40										
Hydrobiosidae	Impact	4	6.38	<0.01	-2.39	0.02	1.87	0.07	3.42	<0.01	-0.83	0.41
	Residuals	40										
Conoesucidae	Impact	4	10.49	<0.01	-2.01	0.05	2.48	0.02	3.90	<0.01	-0.09	0.93
	Residuals	40										
Leptoceridae	Impact	4	20.54	<0.01	-3.91	<0.01	-8.31	<0.01	-1.97	0.06	-8.04	<0.01
	Residuals	40										
Hydropsychidae	Impact	4	15.64	<0.01	-0.45	0.66	2.69	0.01	2.89	<0.01	1.86	0.07
	Residuals	40										

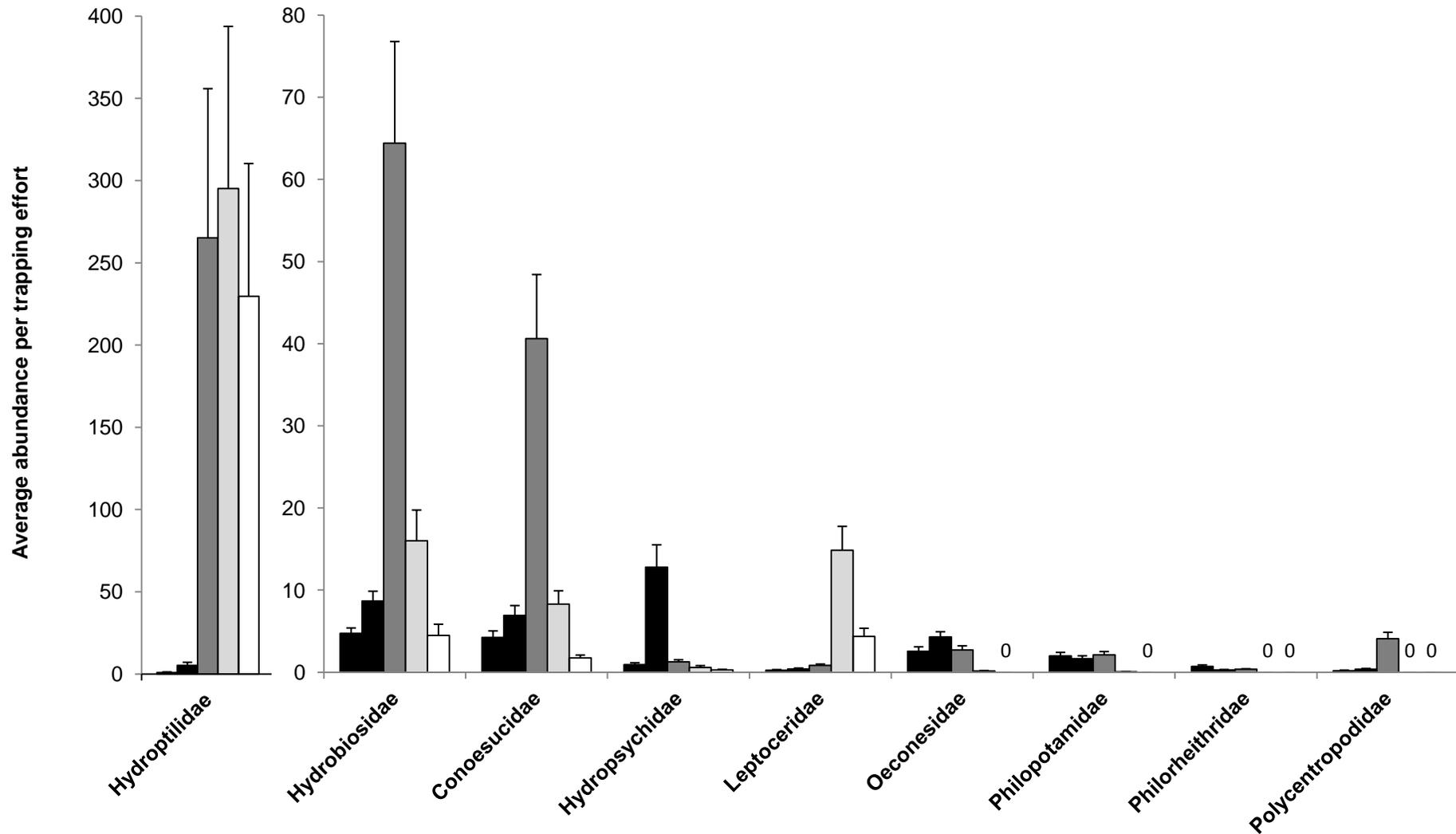


Figure 4. Average abundance with standard bars of adult Trichoptera from the nine families captured using light traps at the five sampling sites that are colour coded to show habitat quality. Grey scaling from open to solid represents the habitat continuum from poor to excellent. Poor habitat quality, high impact area of Riversdale Stream (open bars), marginal habitat area of Riversdale Stream (pale grey), and the reference sites ranging from good (Reference Three; darker grey) and excellent habitat (black). Zeros indicate none of this family were caught at this location.

At all sites, there was more adult diversity at the family level than in corresponding larval populations (Figure 5). In addition, there was evidence for sources of external colonisation to Riversdale Stream because Oeconesidae and Philopotamidae adults were captured at the low impact site, despite not ever being observed as larvae in the stream. A MANOVA indicated a significant interaction between life-stage (adult or larvae) and impact (high, low or reference) across families, suggesting the impact of in-stream habitat quantity generally depended on life-stage of the insect (Table 2.). However, two-way ANOVAs testing families separately indicated that the interaction was primarily associated with two families Hydrobiosidae and Conoesucidae, while Leptoceridae and Hydropsychidae only affected by the level of habitat impact, and Hydroptilidae was affected by both life-stage and impact independently (Table 2.). Next, I examine these patterns of abundance in detail.

Hydroptilidae is known as a pollution-tolerant Caddisfly family with a tolerance score of two (Stark 1993), and this family had no significant interaction between life-stage and the level of impact (Table 2.). However, both of these factors significantly affected Hydroptilidae independently, suggesting abundances of each life-stage were affected similarly at the different impact levels despite the abundance of adults being greater than that of larvae (Table 2.). Moreover, there were large differences in abundance between Riversdale and combined reference sites, due to the very high abundances of this family. Specifically, abundances of this family were substantially higher in the grassland sites (Riversdale Stream and Reference Three) in comparison to the more pristine reference sites where abundances were very low for both life-stages.

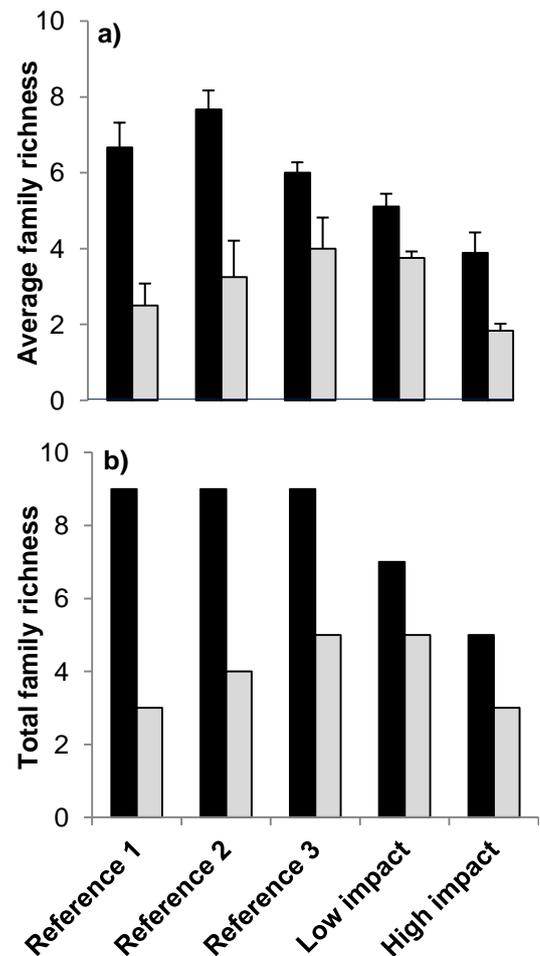


Figure 5. Average (and standard error) (a) and total (b) richness out of nine possible families for adult (black bars) and larval Trichoptera (grey bars) at the different sampling locations. Riversdale larval sampling sites were separated into high (three sites) and low impact (2 sites) based on habitat scores calculated using a modified New Zealand habitat assessment standard from Harding *et al.* (2009).

Table 2. Using the five most common Trichoptera families a MANOVA and then multiple univariate two-factor ANOVA's (abundances log-transformed) were completed with additional contrasts. Contrast levels were high impact against low impact (Riversdale sites), reference (all three streams) against Riversdale (both impacts) and then reference against high and low impacted sites separately. The five Riversdale Stream larval sites separated into high (3 sites) and low impacted (2 sites) because of their respective poor and marginal habitat scores calculated using a modified New Zealand habitat assessment standard from Harding *et al.* (2009). *F ratio for the MANOVA are an approximate calculated by R. The Pillai value for impact, stage and the interaction are 2.48, 0.66 and 0.81, respectively.

		ANOVA			Riversdale against reference sites		High impact Riversdale against reference sites		Low impact Riversdale against reference sites		High impact against low impact Riversdale	
		df	*F	P	t	P	t	P	t	P	t	P
MANOVA	Impact	4	24.91	<0.01								
	Stage	1	28.11	<0.01								
	Interaction	4	3.84	<0.01								
	Residuals	77										
Hydroptilidae	Impact	4	25.28	<0.01	-6.46	<0.01	1.00	0.32	-6.46	<0.01	3.72	<0.01
	Stage	1	6.69	0.01	2.02	0.05						
	Interaction	4	0.77	0.55	-0.37	0.71	1.37	0.18	-0.08	0.93	-0.98	0.33
	Residuals	77										
Hydrobiosidae	Impact	4	29.13	<0.01	4.32	<0.01	5.32	<0.01	5.62	<0.01	-7.92	<0.01
	Stage	1	7.61	<0.01	0.64	0.52						
	Interaction	4	3.05	0.02	-2.31	0.02	-0.65	0.52	-2.53	0.01	2.24	0.03
	Residuals	77										
Conoesucidae	Impact	4	38.8	<0.01	3.13	<0.01	8.95	<0.01	5.19	<0.01	-10.37	<0.01
	Stage	1	57.9	<0.01	-7.14	<0.01						
	Interaction	4	10.1	<0.01	-0.09	0.93	-5.12	<0.01	-1.21	0.23	4.72	<0.01
	Residuals	77										
Leptoceridae	Impact	4	30.7	<0.01	-9.12	<0.01	7.09	<0.01	-7.89	<0.01	0.11	0.92
	Stage	1	0.02	0.88	-2.12	0.03						
	Interaction	4	2.4	0.06	1.01	0.32	-2.93	<0.01	0.40	0.69	1.93	0.06
	Residuals	77										
Hydropsychidae	Impact	4	31.18	<0.01	4.30	<0.01	1.75	0.08	4.83	<0.01	4.67	<0.01
	Stage	1	0.35	0.56	-0.10	0.92						
	Interaction	4	1.46	0.22	0.32	0.75	-0.77	0.44	0.17	0.87	0.47	0.64
	Residuals	77										

Both Hydrobiosidae and Conoesucidae families had a significant interaction with life-stage and impact level (Table 2.). Of these families, no larvae and very few adults were recorded in the high impact site of Riversdale Stream, which may have strengthened this interaction. However, adult Hydrobiosidae were more abundant than larvae at Riversdale Stream suggesting a lack of colonists should not have limited larval populations. Moreover, Conoesucidae were the only family that was more abundant as larvae than as adults, and the abundance of larval stages at the low impacted site was similar to that of the pristine reference sites.

Overall, Leptoceridae and Hydropsychidae populations were significantly affected by the level of impact across the sites suggesting little difference in abundance between life-stages (Table 2.). The Riversdale populations were similar to one another for both families, and greater than all Reference sites for Leptoceridae (Table 2.). However, the Hydropsychidae population was substantially greater at Reference Two, whereas all other sites had much smaller populations (Table 2.).

Oviposition

The average abundance of egg masses found per reach was significantly higher for reaches with added boulders than reaches with only natural substrate (Table 3., Figure 6.), indicating that adding oviposition habitat increased oviposition. However, the lack of oviposition in the highly impacted area indicated this response was dependent on habitat condition (Figure 6). Thus, availability of oviposition habitat may be limiting some (less impacted areas) parts of Riversdale Stream, however there was another limiting factor at Riversdale Stream restricting the recovery of sensitive insects via oviposition.

Table 3. Nested ANOVA on the average abundance of egg masses on substrate type in habitats with either added boulders or naturally occurring large cobbles within two Riversdale sites in the low impacted areas of the stream. Impactedness was determined by a modified New Zealand habitat assessment standard from Harding *et al.* (2009).

	df	MS	F	P
Error: Site				
Residuals	1	3.18		
Error: Within				
Substrate Residuals	1 37	5.18 0.57	9.13	<0.01

DISCUSSION

There is no question that reversal of habitat loss must form an important component of restoration (Bond and Lake 2003), but a number of other factors clearly also influence local populations. I discovered that Riversdale Stream had a depauperate in-stream fauna, whereby poor stream habitat condition had a strong negative relationship with larval diversity. This habitat quality gradient also encompassed habitat complexity, and therefore one may have assumed that the addition of habitat to Riversdale Stream would allow for the return of species. However, we know that this assumption is not reliable (Palmer and Filoso 2009).

The aim of my study was to determine the likelihood of species returning

to this stream after habitat restoration efforts. Firstly, this meant determining if Riversdale Stream had a connection to the regional species pool. My investigation of Trichoptera families, that made up the populations at surrounding reference streams, indicated these streams had the potential to connect Riversdale Stream to a diverse source of colonists. However, colonisation to Riversdale Stream and other similar streams will principally depend on the distance to, dispersal ability, and the abundance of the source population (Bond and Lake 2003, Brederveld *et al.* 2011). Winterbourn *et al.* (2007), in a study area very close to mine, found female caddisflies could disperse hundreds of metres from the nearest known larval habitat possibly in search of suitable oviposition habitat. In addition, Bunn and Hughes (2007) suggested a successful in-stream population may arise with the dispersal and oviposition of a few individuals. Without considering the effect of dispersal barriers or local landscape features, these studies support the general likelihood of colonisation to Riversdale Stream.

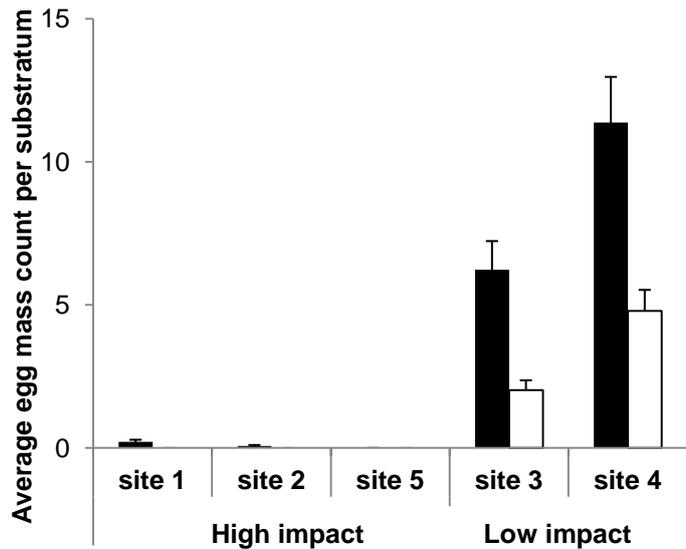


Figure 6. Average (and standard error) abundance of egg masses found on each substratum from five Riversdale sites, separated into high and low impacted because of their poor and marginal habitat scores, respectively, calculated using a modified New Zealand habitat assessment standard from Harding *et al.* (2009). Egg masses were counted from added boulders, (solid bars) and naturally occurring large cobbles, (open bars) from the manipulation and control reaches respectively of each site.

Based on other projects, the most abundant adult caddisfly families caught in my study (Hydroptilidae, Hydrobiosidae, Conoesucidae, Hydropsychidae, and Leptoceridae) are unlikely to attract restoration efforts, but the presence or return of families missing from Riversdale Stream however, are more likely to be a focus, and their return a determinant of restoration success. Additionally, populations of other sensitive insect orders, such as Ephemeroptera (mayflies) and Plecoptera (stoneflies), will also be determinants of restoration success as these orders were largely absent from Riversdale Stream, with the exception of *Deleatidium*, generally a common mayfly, but rare at Riversdale Stream. Populations of adult Trichoptera at Riversdale Stream were different to those in all the surrounding reference streams. Nine Trichoptera families made up the species pool from the reference sites, but Riversdale sites had lower adult diversity, with the high impact site lacking four families of Trichoptera (*Oeconesidae*, *Philopotamidae*, *Philorheithridae* and *Polycentropodidae*), and the low impact site lacking two families (*Philorheithridae* and *Polycentropodidae*). This suggests surrounding streams may add species to Riversdale Stream's species pool. However, this does mean that only two families are not currently getting to Riversdale Stream; and external colonisation is occurring because some families captured as adults (*Oeconesidae* and *Philopotamidae*) were not present as larvae.

Secondly, I was interested in whether the missing families had particular habitat requirements for different life-stages, and whether there was potential to enhance or add this habitat at Riversdale Stream. Here, I discovered that a combination of in-stream and oviposition habitat was likely restricting the return of sensitive species that might occupy a grassland stream such as, *Oeconesus* (*Oeconesidae*) or *Polyplectropus* and *Plectrocnemia* (*Polycentropodidae*). Riversdale Stream would appear to require additional habitat, such as areas of vegetation that produces inputs of detritus, shade for moss, algal growth and reduction of macrophytes, as well as cooler water temperatures to support populations of these invertebrates. The specific habitat requirements of the missing larvae indicate (Table 4.) that currently, Riversdale Stream does not have the habitat capacity to support these families as larvae regardless of their colonisation ability. Particularly for the likes of forest specialists that are highly unlikely to occupy non-forested habitats such as *Zealandopsyche* (*Oeconesidae*), a beech forest specialist. In addition, variations in substrate size (fine sediment removal) and stream flow (channel narrowing) would be needed, as well as general habitat improvements, if this stream were to support these highly sensitive species

Table 4. Detailed taxonomy and in-stream habitat requirements of the taxa missing as larvae from Riversdale Stream that were found in the surrounding reference sites. Macroinvertebrate Community Index (MCI) indicates level of pollution sensitivity on a scale from one to ten, with a score of ten indicating strong sensitivity to pollution.

Family	Genus	Species	MCI (Stark 1993)	Aquatic habitat preferences and requirements (Cowley 1978, Winterbourn <i>et al.</i> 2006)
Oeconesidae	<i>Oeconesus</i>	<i>O. maori</i> <i>O. similis</i>	9	Small bush or forested streams with large accumulations of leaf litter or mosses on which they are found in and feed on. Their cases are built from leaf or stone depending on species. Larvae require clean water streams with plenty of flow, although larvae are found in pools with detritus or silty gravels. Pupation can occur on the underside of large rocks or where the larvae buries itself in gravel substrate.
	<i>Zealandopsyche</i>	<i>Z. Ingens</i>	9	Beach forest streams where they live amongst beech litter accumulations at the bottom of pools or slow flow areas. Larvae build neat cases from plant fragments, leaves (early instars) or twigs (late instars).
Philopotamidae	<i>Hydrobiosella</i>	<i>H. stenocerca</i>	9	Larvae feed mostly on detritus and diatoms but not filamentous algae, and occupy stones in riffles of small-forested streams, sometimes found in larger rivers or open streams in grasslands in the northern south island.
Philorheithridae	<i>Philoreithrus</i>	<i>P. agilis</i>	8	Cold water forested streams where they build stony cases
Polycentropodidae	<i>Polyplectropus</i>	<i>P. puerilis</i>	8	Occupy nets attached to either rocks or branches and with gravels, sticks and often silt. Common on quietly flowing parts of streams and lakes.
	<i>Plectrocnemia</i>	<i>P. maclachlani</i>	8	Similar to <i>P. puerilis</i> however, are often found in faster currents and also found in larger rivers and requires the undersides of rocks for pupation.

Acknowledging the habitat needs for all target species in restoration project goals is clearly important (Lake *et al.* 2007, Muotka and Syrjänen 2007), and integrating this knowledge into goals is essential, as all desired species cannot be assumed to just recolonise (Hughes 2007). Therefore, when considering restoration, the habitat requirements of possible target families (from Table 4.) gives an indication of what habitat properties are potentially required in Riversdale Stream, and a lack of these may very likely restrict species recovery. Riversdale Stream's restoration potential is also important to consider and should not be expected to mimic that of a forested stream. If habitat restoration of this stream is successful, colonisation of forest dwelling species, specifically species that may only occur in beech forests or species that prefer not to disperse across non-forested habitat, is unlikely. Habitat connectivity, such as dispersal corridors, could however increase this likelihood, but consideration of appropriate target species, such as grassland or open bush dwelling species, is more probable to ensure success.

Therefore, thirdly I investigated whether there were barriers to colonisation and if those barriers could be overcome by creating or enhancing connectivity. Local landscape features and environmental gradients are known to greatly influence flight and dispersal behaviour of aquatic insects (Bohonak and Jenkins 2003). The distribution of adult aquatic insects is strongly influenced by vegetation densities with important differences between grassland, scrubland, and forest habitats detected (Winterbourn *et al.* 2007, Didham *et al.* 2012). In addition, some aquatic insects are restricted to flight only in particular vegetation types such that certain groups were only found flying in forested or grassland vegetation, whereas others occurred within both kinds of vegetation. Specifically, Winterbourn *et al.* (2007) found that of 38 caddisfly taxa, eight were excluded from flight in grasslands and five from forested environments. Additionally, forest dwelling species of other orders, such as Plecoptera, have also been found to generally not disperse far beyond the riparian zone (Winterbourn *et al.* 2007). As Riversdale Stream is currently surrounded directly and only by grassland vegetation, this possibly already excludes certain species. Therefore, without consideration of the habitat needs of all life-stages of target species, recovery may be slow or incomplete (Jansson *et al.* 2007, Muotka and Syrjänen 2007). Therefore, the need to discover and realise the importance of surrounding landscape features as a barrier to certain species will also be important for determining goals for restoration. In addition to in-stream and surrounding habitat, other factors can influence colonisation, such as population abundance and propagule pressure specifically (Hughes *et al.* 2009, Brederveld *et al.*

2011). Oeconesidae and Philopotamidae adults but not juveniles, were found at Riversdale Stream, indicating they must have arrived via flighted dispersal. If population abundance is in fact important to the likelihood of dispersal success (Brederveld *et al.* 2011), and we ignore the constraints of degraded in-stream habitat momentarily, these families may have approximately the minimum abundance (between 5-10 individuals found per trapping effort) required to be likely to disperse such a distance. In comparison, Polycentropodidae and Philorheithridae were caught in fewer numbers and possibly below this abundance threshold (1-3 individuals per trapping effort), suggesting that these two families may have low probability of colonising Riversdale Stream even if habitat conditions in- and connectivity to- Riversdale Stream were improved.

Finally, if all of the above problems were solved and all desired groups eventually arrived, a lack of oviposition habitat may limit repopulation at Riversdale Stream. If oviposition site selection has preference, due to growth or survivorship within early developmental stages, this can have strong implications for population dynamics (Hoffmann and Resh 2003, Winterbourn *et al.* 2007). It has been suggested that species with larvae that display specific dietary needs may exhibit a more highly developed oviposition strategy (Reich and Downes 2003). Thus, it is possible that egg-laying decisions made by adults might benefit their young by providing a suitable habitat in which to develop. For example, within the Hydrobiosidae family, different species select emergent substrate depending on their surrounding flows (Reich and Downes 2003, Lancaster *et al.* 2010). These environmental factors may influence the hatching success or larval growth of Trichoptera, as it does in odonates, or potentially increase the probability of adult survival and oviposition success (Siva-Jothy *et al.* 1995, Reich and Downes 2003). If desired species oviposition preferences do not coincide with available habitat conditions this can limit their return.

Oviposition habitat is unknown for many aquatic insect groups, therefore, we are unable to determine possible oviposition limitation for other groups and in particular those of desired species. However, oviposition habitat often includes larger cobbles and boulders, logs, or overhanging vegetation, which can be relatively unavailable in degraded streams such as Riversdale. With the addition of known Hydrobiosidae oviposition habitat (emergent boulders), the average abundance of egg masses was substantially higher compared to unmodified habitat. This suggests oviposition in the less impacted areas at Riversdale Stream could be increased by the addition of suitable habitat. However, the lack of oviposition even when boulders were added in the highly impacted area indicates

adding oviposition habitat could only increase egg mass abundance if there was not another limiting factor. The great depths of sediment in the highly impacted areas meant the addition of emergent substrate had relatively little long-term effects on the availability of oviposition habitat because even though boulders emerged above the water level, their undersides sank into the sediment making them unavailable for oviposition. A similar result was also found in a study of a high sediment urban stream where larger substrates were also highly embedded (Blakely *et al.* 2006). Therefore, the lack of oviposition habitat can severely limit colonisation events in these areas and numbers of early instar larvae Hydrobiosidae populations are likely limited by availability of oviposition habitat.

However, due to density-dependent effects on first star larvae, addition of oviposition habitat to Riversdale Stream may still have little impact on future populations if habitat for late instars is also insufficient. This is supported by an experimental manipulation of multiple river stretches in south-east England, where 92 % of dobsonfly egg masses were removed from a reach, and added into a recipient reach (effectively almost removing or doubling egg mass abundances, respectively). Due to density dependent effects this resulted in no noticeable differences of later populations as there was insufficient habitat to support such large numbers of early in-star larvae (Hildrew *et al.* 2004). This is likely to be the case at Riversdale Stream where the addition of oviposition habitat may increase abundance of egg masses and early in-start larvae however, inadequate quantities of suitable habitat caused by excessive sedimentation are likely to limit populations in this case. Therefore, habitat needs to be improved so Riversdale Stream can support greater invertebrate populations.

Riversdale Stream inhabitants may also benefit not only from improving in-stream habitat and riparian conditions directly beside the stream but also from an extension of the riparian zone to create dispersal corridors. This will connect Riversdale Stream to the surrounding national park to aid the colonisation of specialist or sensitive species as land-use change can be the dominating factor structuring adult aquatic insect distribution at a landscape scale (Didham *et al.* 2012). Although Riversdale Stream has a depauperate in-stream fauna and degraded habitat, Riversdale also has the potential to be connected to a diverse species pool because of the close proximity of high quality diverse stream habitat. This potential connection puts Riversdale Stream in the best possible location for a restoration project to take advantage of the desired response and reduce any time lags for restoration. As Huxel and Hastings (1999) found, a restoration site close to a species pool had significantly reduced response times, compared to restoration at random.

Realistic and more short-term goals for habitat restoration of Riversdale Stream and other streams in a similar situation should include enriching and expanding the habitat for families that already occur in the stream. For example, the habitat requirements of the family Conoesucidae; *Pycnocentria*, are found in slower parts of streams with filamentous algal, moss, watercress and decaying vegetation providing there is clean water (Cowley 1978). *Olinga* are generally found in moderate to fast flowing cobble streams and feed on detritus similar to *Pycnocentrodus* (Cowley 1978). Moreover, Hydrobiosidae habitat requirements may also be more realistic and achievable to enhance in the short-term for Riversdale Stream. Six genera and upwards of 100 described species make up this family and 44 % of the Caddisfly fauna of which, two genera were found in the Riversdale Stream larval community. General habitat requirements for this family are either open or shaded streams with moderate to fast flows in stony streams. Additionally, non-forest specialists from the family Oeconesidae (not found as larvae in Riversdale Stream), for example the genera *Oeconesus*, prefer bush-covered streams that receive leaf litter. However, the pressing limitation of sedimentation and the addition of riparian vegetation will be a priority for restoration efforts in this stream before these species are likely to colonise (Winterbourn *et al.* 2006).

In this study, I used family level identification of adult Trichoptera insects. Finer taxonomic resolution can offer outcomes that are more detailed, however, higher taxonomic resolution was probably unnecessary due to the similar general habitat requirements of interesting groups, such as described in the overview in Table 4. In addition, identifying to family level is a much more cost effective perspective when considering additional information that could be supplied and the level of specific species information available, particularly when considering the practical use for restoration managers (Törnblom *et al.* 2011). The importance of this work was to determine a connection between potential restoration sites with surrounding colonisation and species sources as this would play an important role in recovery and something that should be considered in every restoration project.

Oviposition studies suggest adult's egg-laying behaviour restricts oviposition in unsuitable egg development or larvae habitat (Reich and Downes 2003, 2004, Blakely *et al.* 2006, Lancaster *et al.* 2010), Winterbourn *et al.* (2007) proposed that some species may not disperse out of their preferred habitat type, and finally Cowley (1978) describes larvae specific habitat preferences. Therefore, families missing from Riversdale or similar streams are not likely to colonise currently due to inadequate in-stream and surrounding habitat. Consequently, it can be concluded that Riversdale

Stream's sensitive insect population is currently limited by poor in-stream habitat. However, it would be wrong to assume that improving habitat will result in the return of all species. If Riversdale Stream is considered for restoration, both habitat within and around the stream will need to be improved so it can support all life-stages of the desired species or groups. Even then, we know the four missing families from Riversdale are highly sensitive and the majority would require riparian vegetation that created "forest like" in-stream conditions.

Conclusion

A connection to a healthy species pool is likely to be a strong determinant for biotic recovery success in restoration and this can be influenced by the abundance of species or groups, whereby the greater the abundance the more likely a colonisation event will be. In addition, some species are not likely to actively travel out of their desired habitat, for example forest-dwelling species may require corridors between restored and intact habitats. Consequently, this has important implications, particularly limitations, for choosing appropriate target species; rare and habitat-sensitive species may be desirable but they may take longer to arrive at a restoration site and creating habitat suitable for all life-stages may not be feasible or appropriate. Therefore, I urge restoration planners to be realistic when considering desired species and priority for restoration should be given to streams with a connection to a diverse species pool. My research indicates the reversal of habitat damage and loss is important and we must continue to develop techniques to reverse this damage. A community, such as Riversdale Stream, remaining connected to an intact regional species pool is expected to show a significant positive response to habitat restoration, in contrast to a community that is not as well connected. Thus, while improved habitat is an important part of the restoration process, my study indicates aspects of the surrounding environment need to be considered when planning a restoration project.

CHAPTER THREE

Is active sediment removal an option when fine sediments limit invertebrate recovery in stream restoration?



A splash of colour on a grey day, at Riversdale Stream

INTRODUCTION

Globally, urban and agricultural influences have degraded ecological conditions of many streams and rivers (Roni *et al.* 2008, Vörösmarty *et al.* 2010). Degradation of these environments is associated with run off from roads, drains, and pastures and often includes high levels of fine sediments (inorganic particles <2 mm), nutrients, and other contaminants that can alter water chemistry and degrades habitat quality (Bond and Lake 2003, Palmer *et al.* 2005, Bernhardt and Palmer 2007, Lake *et al.* 2007). Habitat quality, particularly sediment size and heterogeneity, is known to influence invertebrate communities. This is not surprising as substratum and associated heterogeneity influence oviposition, in-stream movement, feeding patterns, provides shelter from predation and refuge from disturbances (Reice *et al.* 1990, Palmer *et al.* 1992, Dole-Olivier *et al.* 1997, Lake *et al.* 2007). Individual invertebrate species have different habitat preferences and limitations, and consequently a community present in a habitat should reflect its environmental state. Therefore, invertebrates are an important and commonly used tool for assessing changes in biotic habitat quality (Quinn and Hickey 1990, Stark 1993).

Stream restoration (i.e. the return of a degraded ecosystem to a healthier state, Chapter one) projects generally aim to improve aquatic communities, via replanting riparian vegetation and through reintroduction of substratum and flow heterogeneity (Brooks *et al.* 2002, Miller *et al.* 2010). Vegetative buffer strips (or riparian zones) are widely encouraged as an effective technique to protect rivers and streams from negative impacts, such as sedimentation (Lowrance *et al.* 1984, Dillaha *et al.* 1989). The main functions of riparian zones are erosion control, buffering against nutrient and sediment runoff, moderation of shade and water temperature, introduction of detrital subsidies and enhancing in-stream habitat, and nitrogen processing (Lowrance *et al.* 1984, Dillaha *et al.* 1989, Craig *et al.* 2008). The assumption that improving habitat features increases species richness and abundance is often referred to as the 'field of dreams hypothesis' (Palmer *et al.* 1997). Although habitat restoration of certain type, such as riparian zones may be necessary, it appears to be insufficient in many restoration projects to significantly recover benthic communities (Bond and Lake 2003, Alexander and Allan 2007, Miller *et al.* 2010, Palmer *et al.* 2010).

Sedimentation, one of the key influences on substrate, is the process whereby fine sediments settle out of the water column having the potential to deposit in large quantities on the streambed. This can occur directly or indirectly through tillage, deforestation, urbanisation, water abstraction and

mining (Taylor *et al.* 2004, Bruns 2005, Kreuzweiser *et al.* 2005, Larsen *et al.* 2009, Matthaei *et al.* 2010). In particular, when livestock have access to stream banks they cause erosion, slumping, and pugging of the surrounding land allowing fine sediments to enter the water (Trimble and Mendel 1995, Braccia and Voshell Jr 2007, Carline and Walsh 2007). Furthermore, sediment build-up has been identified as one of the most important non-point sources of pollutants in streams and is getting worse as catchments become increasingly modified and intensified (Lenat *et al.* 1981, Wood and Armitage 1997, Matthaei *et al.* 2006, Larsen *et al.* 2011).

Sediment inputs affect stream communities through a variety of processes. These changes to in-stream habitats, particularly substratum characteristics, reduction and smoothing of habitat, and reduced access to trophic resources have corresponding impacts on benthic community assemblages (Lenat *et al.* 1981, Rae 1987, Erman and Ligon 1988). There is also evidence that riparian zone benefits may be hindered in the presence of high sediment loads. For example, significantly lower breakdown rates of wood and leaves have been observed when associated with high sediment cover (Tank and Winterbourn 1996, Spänhoff and Arle 2007, Young *et al.* 2008). Moreover, high levels of benthic sediment can act as a nutrient reservoir, which is released during disturbance, despite riparian protection restricting further nutrient inputs (Angradi 1999). Furthermore, the basis of the aquatic food chain (algae and detritus) is often smothered by sediment covering the stream bottom. Therefore, sedimentation can limit the capacity of streams to support a diverse fauna, often resulting in the exclusion of pollution-sensitive taxa (Owens *et al.* 1996, Collier *et al.* 1997). Only a few groups, such as Oligochaeta, Amphipoda, and Ostracoda may benefit from the accumulation of sediment (Dance and Hynes 1980, Quinn *et al.* 1997, Collier *et al.* 1998, Delong and Brusven 1998, Bo *et al.* 2007). Predominantly, sedimentation decreases the abundance of desirable sensitive invertebrate populations, such as Ephemeroptera, Plecoptera and Trichoptera (EPT) and a reduction in overall diversity via habitat bottle necking (Gray and Ward 1982).

Restoration projects that measured natural removal of deposited sediment with no active interference have had mixed results. For example, one of two stream sections fenced with 3 m grass riparian buffer showed reductions (averaged 6 % decline over sites) in sediment cover over three years of observations in Cedar Run Basin, central Pennsylvania (Carline and Walsh 2007). The remaining sections further increased in sediment cover (from 32 % to 52 %). Carline and Walsh (2007) suggested the increase in sediment was due to slower flows and sources in unfenced areas

upstream of the restoration site, however, further sediment build-up was observed in areas with the greatest overall degradation. In addition, Greenwood *et al.* (2012) found that riparian management had not brought about reductions of deposited sediment or increases of water velocities in Canterbury Plains streams and suggested highly sedimented streams were unable to naturally remove sediment, restricting recovery of sensitive invertebrate populations. Thus, there is a need to develop sediment removal protocols to aid highly degraded areas in sediment removal.

Adding riparian buffer strips may prevent stream conditions from worsening further, but may be insufficient to result in sensitive invertebrate or stream recovery in the face of already high sediment build-up (Parkyn *et al.* 2003). Therefore, the effect of restoration efforts using riparian plantings are likely to be limited in the presence of already high levels of in-stream sediment, particularly, when these streams also have slow water velocity and poor in-stream habitat structure that restricts natural sediment removal (Greenwood *et al.* 2012). This situation reflects a stream in hysteresis, where the pathway to recovery differs from that to degradation. A stream with heavy sedimentation may be caught in a state of hysteresis and will have to cross one or many environmental thresholds (on the recovery pathway) to be pushed out of the degraded state and into a restored one (Suding and Hobbs 2009). Moreover, the pathway to recovery will very likely require more than just the removal of the cause of sedimentation (i.e. livestock access) for recovery. For example, actively flushing sediment, as seen in controlled reservoir water releases to mobilise stored sediments (Petticrew *et al.* 2007) may also be required.

To understand why stream restoration projects often fail to recover aquatic communities, we need to identify the limiting factors. Once these are discovered, more effective and targeted restoration techniques can then be designed to push stream conditions out of their degraded state. For example, if reduced flow and sedimentation are limiting recovery and can be mitigated, it is expected that a positive response to other restoration techniques such as riparian zones may be seen. Active sediment removal (e.g. via water flushing) may be a viable option for streams in this condition, though to my knowledge, this option has not been widely considered.

In this study, I firstly determined if high-quantities of fine sediment were limiting the recovery of sensitive species, by adding high-quality (sediment-free) habitat patches to a stream severely affected by sedimentation and measuring community responses. Secondly, active sediment removal methods were trialled using a combination of techniques including increasing stream velocity and active

removal of sediment from the streambed with flushing to attempt to push sections of stream out of the heavily sedimented stable state. This was to determine if active sediment removal is an option for stream restoration. I hypothesised that due to the hysteresis effect of sedimentation, the removal of fine sediment from the stream would result in the recovery of sensitive invertebrates, and that the combination of mechanical and hydrological movement of sediment would be most effective.

METHODS

Rock-basket experimental setup

Rock-baskets were placed in the same experimental manipulation sites as in the oviposition experiment from Chapter 2. The baskets (30 x 20 x 5 cm) were made from wire mesh (19 mm diameter), and filled with cleaned stones (2-5 cm diameter) (Death 1996, Elser 1999). To reduce sediment build-up, baskets were elevated 5-10 cm above the sediment layer using fencing wire to create legs, ensuring the basket remained as high habitat quality for the entire duration of the study.

Ten rock-baskets per site were placed randomly across the stream, approximately 1 m apart within each 10 m site. These were then left for four weeks in February 2011 before collection. On collection, four Surber samples per site were also taken to make comparisons between communities in added and natural habitat. Surber samples (0.06 m²; 250 µm mesh) were collected at approximately 2 m intervals in random locations using standard protocols (Harding *et al.* 2009), and stored in 70 % ethanol until identified later in the laboratory. Benthic invertebrates were identified to genus using keys in Winterbourn *et al.* (2006) at a 10-30 x magnification under a dissection microscope.

Sediment removal experimental setup

Five 24 m sections of Riversdale Stream were selected with the greatest consistency in width, flow type (run, riffle or pool), and sediment depth as blocks in the sediment manipulation experiment to minimise natural variation within and among blocks. Blocks were downstream of the Mount White Road bridge and the rock-basket experimental sites and at least 20 m apart. Experimental setup was completed during two days in September 2011 when each section within a block was allocated one of four treatment types in a randomised block design with buffer strips between treatments (Figure 1.). The treatments were water-blasting (water-blasted vs. not) and narrowing (narrowed vs. not),

arranged in a two by two factorial design within a block. These manipulations were chosen to remediate the specific causes of sedimentation for this stream, i.e. physically removing sediment and increasing water velocity. To remove sediment from the streambed water-blasting was applied, and narrowing was intended to increase water velocity allowing for natural movement of sediment. The buffer strip between was used to reset flow changes between treatments. Pre- and post-treatment measurements were collected for stream width, depth, velocity (using a Marsh McBirney flow meter), and sediment depth. Sediment depth was also recorded at monthly intervals for five months after manipulation.

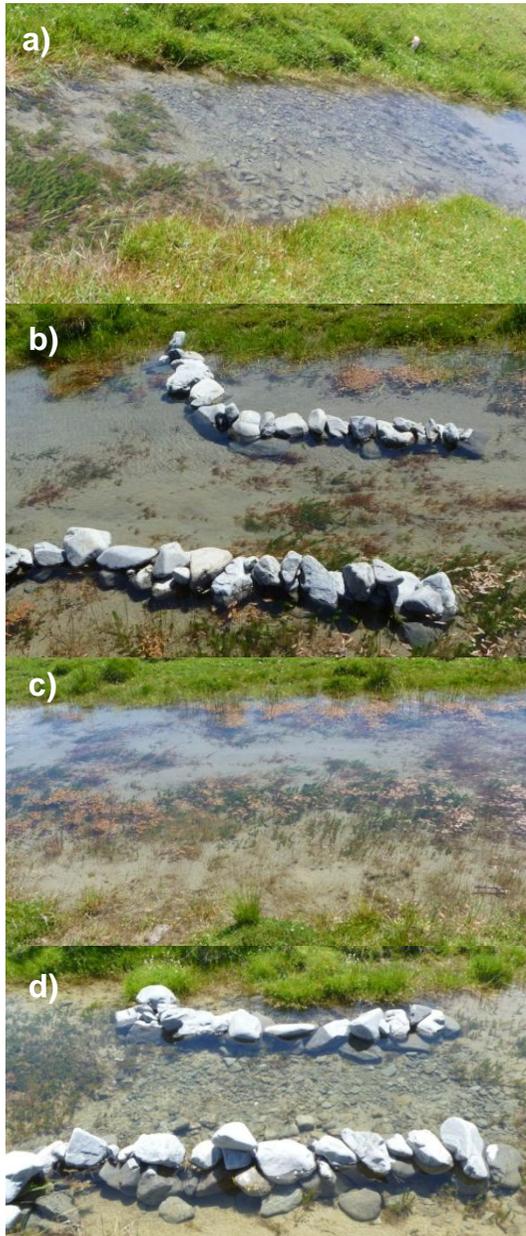
Water-blasting was carried out using a 3000 psi trailerised diesel waterblaster and continued in a treatment section until sediment was cleared or water-blasting exceeded a 1-hour maximum effort time limit. This time limit was reached in blocks Three and Four, which resulted in these blocks not being completely cleared of sediment; these blocks were the widest and had the most heavy sediment deposits. Narrowing was achieved by importing numerous large boulders to each bank within the stream, reducing width to no more than a third of the original channel width or to a minimum of 1 m. Faster velocity was created through the middle of the channel rather than to one side to reduce the risk of further bank erosion.

Invertebrate communities were collected using three Surber samples in each replicate treatment per block in February 2012 (five months post-experimental manipulation). Surber samples (0.06 m²; 250 µm mesh) were undertaken at approximately 1 m intervals using standard protocols (Harding *et al.* 2009) and collected invertebrates were stored in 70 % ethanol for identification in the laboratory. Benthic invertebrates were identified to genus using keys in Winterbourn *et al.* (2006) at a 10-30 x magnification under a dissection microscope.

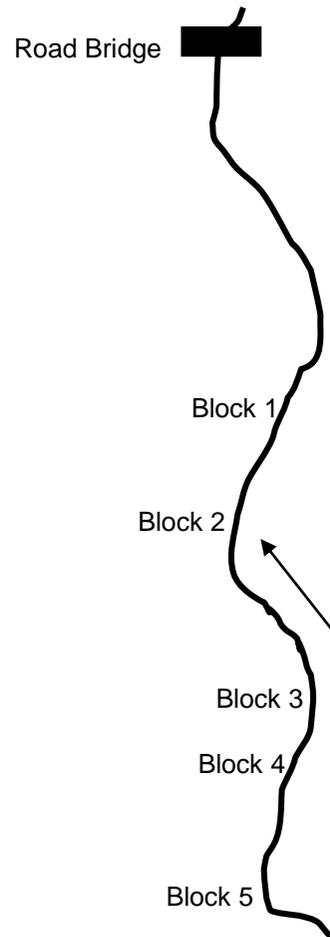
Statistical analysis

I assessed variation in benthic invertebrate community composition using an ordination of species abundance for both the rock-basket and sediment experiment community data. The abundance of each species (square-root transformed to down weigh highly abundant species) was used to create a Bray Curtis dissimilarity matrix which was subjected to ordination using non-metric multidimensional scaling (NMDS) with 20 random starts in both two and three dimensions using PRIMER (v.5; Clarke

a - d) Examples



e) Riversdale Stream



f) Block 2

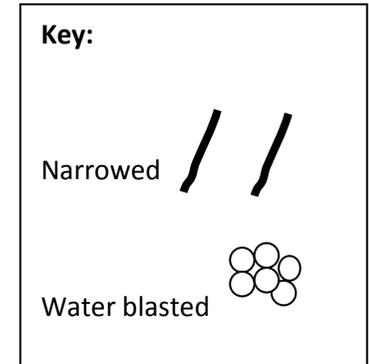
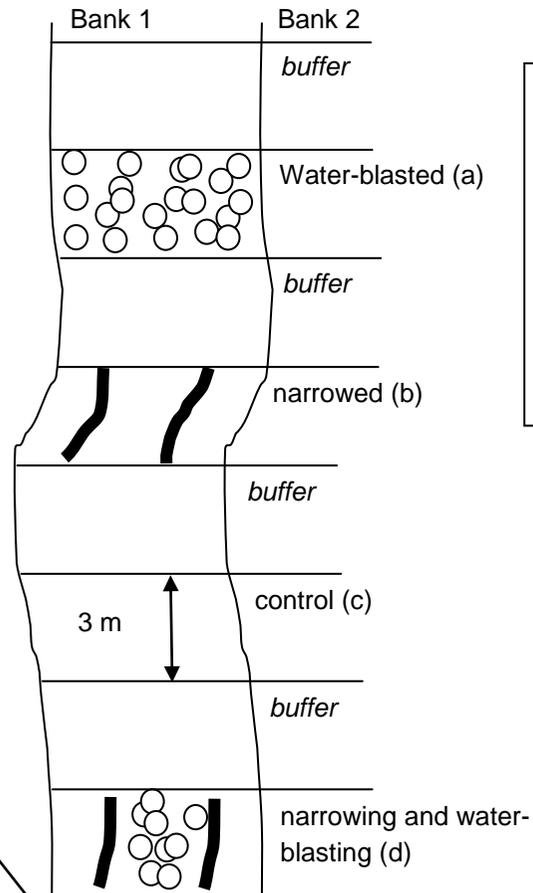


Figure 1. Experimental design of sediment removal experiment at Riversdale Stream (e) with layout of block 2 as an example (f), and photographs of treatments (a-d) post-experimental setup. Treatments were allocated at random in each block, with five replicate blocks at least 20 m apart downstream of the Mount White Road Bridge, and a 3-m buffer strip between treatments. Treatments were: a, water

and Gorley 2000). This produced measures that represented differences in community composition across samples along the ordination axes. I retained two dimensions in the final model to simplify analysis, as adding the third dimension resulted in a negligible decrease in stress. Spread within and among ordination community groups was calculated in R, V.2.15.0 in package *vegan*, using a beta-dispersion function and permutation tests for the rock-basket communities (Crawley 2007).

To further investigate change in community composition in both rock-basket and sediment experiments, I calculated the percentage of EPT and Quantitative Macroinvertebrate Community Index (QMCI), a commonly used metric of New Zealand invertebrate community health (Stark 1993). Rarefied richness, Margalef's diversity, and total abundance were also calculated in PRIMER to assess variation in community structure (Clarke and Gorley 2000). Proportions were arcsine-square-root-transformed and abundance was log-transformed to meet assumptions of normality.

Multiple split-plot ANOVA were run on rock-basket and associated Surber samples with invertebrate community response variables (% EPT, QMCI, rarefied richness, Margalef's diversity and total abundance). A MANOVA was not possible here due to the specific error term required in split plot designs (Crawley 2007). Split-plot analysis was chosen because effect of substrate treatment (rock-basket or Surber sample of natural substrate) was nested within the levels of high and low sediment, with site as the specific error term (Crawley 2007).

A multivariate (MANOVA) and univariate two-way ANOVA's were run for the sediment experiment invertebrate community responses. These tests were carried out to determine effects of treatment levels (water-blasted or not and narrowed or not) on specific community responses. Additionally, multiple repeated measures ANOVA's were carried out on the treatment averages of the pre- and post-sediment experiment for habitat variables, sediment depth, water depth, velocity and width, to evaluate the effect of each treatment on specific response variables over time. Average sediment depths for each treatment were then analysed using linear regression, with collection dates (monthly intervals) as the predictor to further assess trends of sediment depth and changes with time. All analyses were carried out with R, V.2.15.0 (Crawley 2007).

RESULTS

Rock-basket experiment

Natural habitat communities were more tightly clustered than rock-basket communities in the NMDS community composition ordination, suggesting greater community variability among the basket habitats (Figure 2.). The permutation test for homogeneity of multivariate dispersions supported this interpretation with significantly different dispersion among groups (beta dispersal: F -ratio 8.32; P -value, <0.01). In addition sensitive species, EPT, and other invertebrates that score highly on the MCI were associated with the rock-basket communities in the species ordination (Figure 2.).

A split plot analysis of community response variables showed strong interactions affecting all response variables except for % EPT where the effect of substratum type (rock-baskets and natural habitat) was dependent on the surrounding sediment conditions (high or low sediment; Table 1., Figure 3). For example, QMCI was higher for rock-baskets compared to natural habitat but this increase was dependent on the level of surrounding sediment. The increase in QMCI due to improved substratum was greater in the high sediment area, suggesting this more degraded habitat has more to gain in terms of species recovery. For % EPT, however, there was no significant interaction, because there were greater % EPT in rock-basket habitats and in low sediment areas regardless of the other treatment (Table 1., Figure 3.).

Diversity and richness declined from natural to basket habitat but the decline depended on sediment cover. Diversity was greatest in the low sediment natural habitat, suggesting this habitat type had a mixture of both pollution-tolerant and sensitive species, whereas the low sediment rock-basket habitats may have mostly had sensitive species and the natural habitat in the high sediment was dominated by tolerant species resulting in lower diversity in these habitats. Total abundance of individuals also had an interaction with substrate and sediment. In the high sediment areas, abundance was higher in the natural habitat, due to the high abundance of pollution-tolerant species. However, in the low sediment areas the opposite occurred whereby the sensitive species dominated the rock-basket habitat patches in higher densities than found in the natural habitat. This suggests the rock-basket habitats provided higher quality habitat than found naturally. The greater dispersion among the rock-basket habitats in the community ordination was likely caused by the increase in ratio of sensitive to tolerant species.

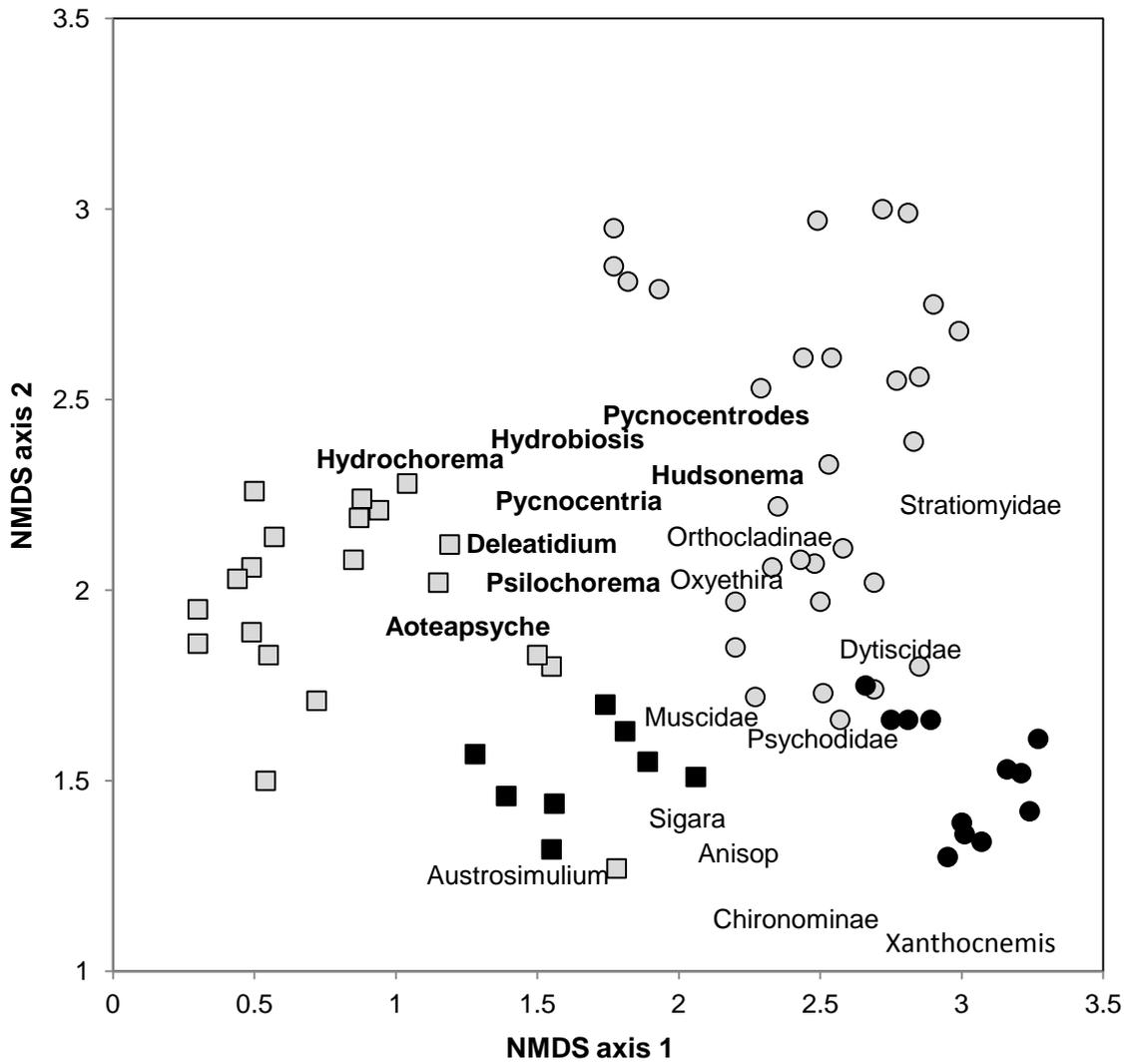
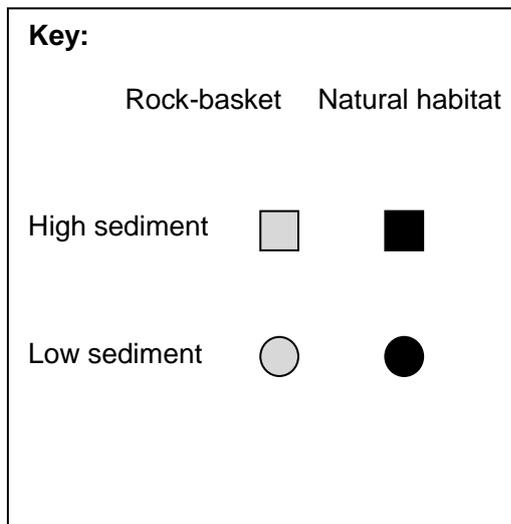


Figure 2. Community (symbols; stress, 0.13) and species (taxa names; stress 0.09) NMDS ordinations on species abundance (square-root transformed) using Bray Curtis dissimilarity matrix with 20 random starts of five sites with four different habitat treatments in Riversdale Stream. Treatments include rock-basket (added habitat) and natural habitat (Surber samples) nested within high or low levels of sediment. Sensitive species are in bold and include EPT insects with a MCI score of five or greater from Stark (1993).



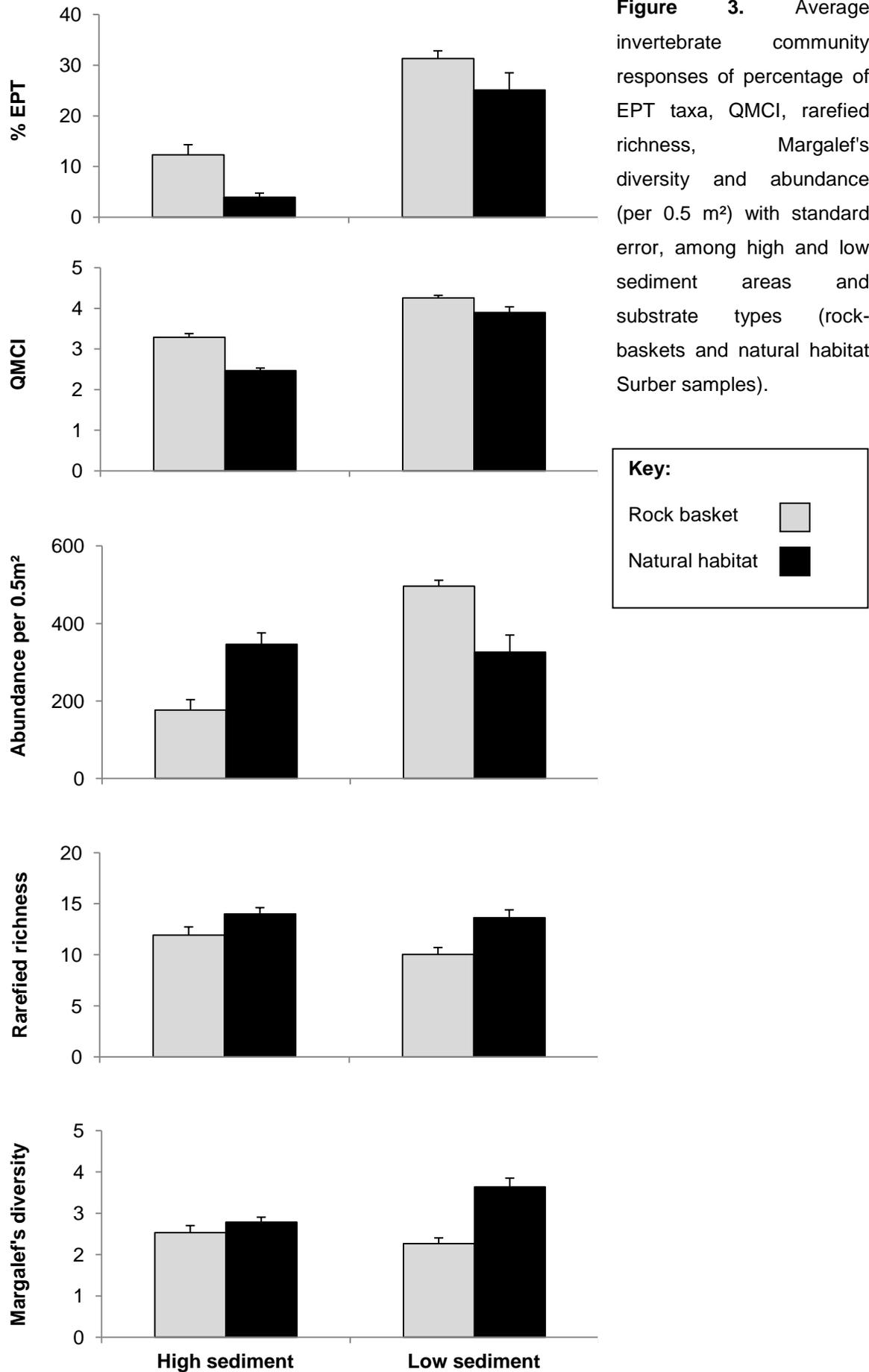


Table 1. Tests for differences in community responses (% EPT, QMCI, richness, diversity and abundance) among substrate treatments (rock-basket and natural habitat) nested within high and low sediment area treatments with split-plot ANOVA, where proportions were arcsine-square-root-transformed and abundances were log-transformed. Site was included as an error term, with levels of the substrate treatment (rock-baskets or natural substratum) nested within either high or low sediment cover areas.

			df	MS	F	P
EPT	AMONG	Sediment	1	1.09	11.62	0.04
		Residuals	3	0.09		
	WITHIN	Substrate	1	0.16	14.61	<0.01
		Sediment x Substrate	1	0.01	0.97	0.33
		Residual	63	0.01		
QMCI	AMONG	Sediment	1	16.83	27.93	0.01
		Residuals	3	0.6		
	WITHIN	Substrate	1	4.83	50.02	<0.01
		Sediment x Substrate	1	1.22	12.66	<0.01
		Residual	63			
Rarefied richness	AMONG	Sediment	1	12.6	1.78	0.27
		Residuals	3	7.07		
	WITHIN	Substrate	1	102.41	21.96	<0.01
		Sediment x Substrate	1	44.48	9.54	<0.01
		Residual	63	4.66		
Margalef's diversity	AMONG	Sediment	1	0.05	0.26	0.64
		Residuals	3	0.2	5.91	
	WITHIN	Substrate	1	7.04	28.69	<0.01
		Sediment x Substrate	1	4.32	17.6	<0.01
		Residual	63	0.25		
Abundance	AMONG	Sediment	1	11.05	4.1	0.14
		Residuals	3	2.7		
	WITHIN	Substrate	1	1.69	9.88	<0.01
		Sediment x Substrate	1	4.62	26.94	<0.01
		Residual	63	0.17		

Sediment removal experiment

In the sediment removal experiment, sediment depth was affected by both water-blasting and narrowing through time, indicated by significant interactions involving time in the repeated measures ANOVA (Table 2.). The rate at which sediment was removed in the manipulation treatments depended on the treatment applied, which in turn was strongly dependent on the amount of surrounding sediment left to be moved (Figure 4.). The reduced width and increased velocity of the narrowed treatment, meant these treatment blocks removed sediment at least twice as fast as sites with water-blasted only (Table 2., Figure 5.). However, sediment levels at the end of this experiment in narrowed treatments had more sediment left to remove than water-blasted sites at the first post-manipulation measurement (Figure 4.). Thus, narrowing the stream channel was effective at removing sediment but slower than water-blasting.

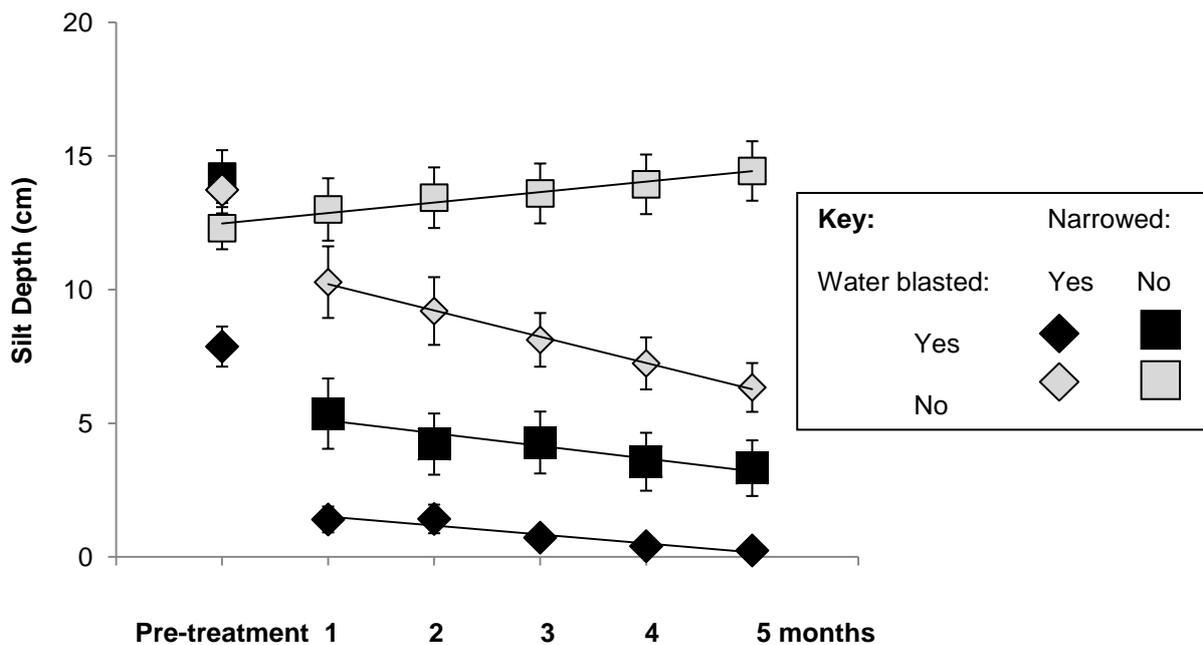


Figure 4. Linear regression of average sediment depth through time (months past experimental manipulation) with standard error bars, among five blocks with treatments applied in a two by two factorial design of water blasting to remove surface sediment and/or narrowing of the channel to increase velocity. Treatment types were: control with no manipulation ($F_{(1,4)} 113.4$, P -value <0.01 , $R^2=0.96$, equation: $y=0.39x + 12.09$), narrowed channel ($F_{(1,3)} 1164$, P -value <0.01 , $R^2=0.99$, equation: $y=-0.98x + 12.17$), water blasted ($F_{(1,3)} 11.79$, P -value 0.02 , $R^2=0.85$, equation $y=-0.47x + 6.04$), and a combination of both water blasted and narrowed channel ($F_{(1,3)} 32.7$, P -value 0.01 , $R^2=0.89$, equation: $y=-0.33x + 2.17$).

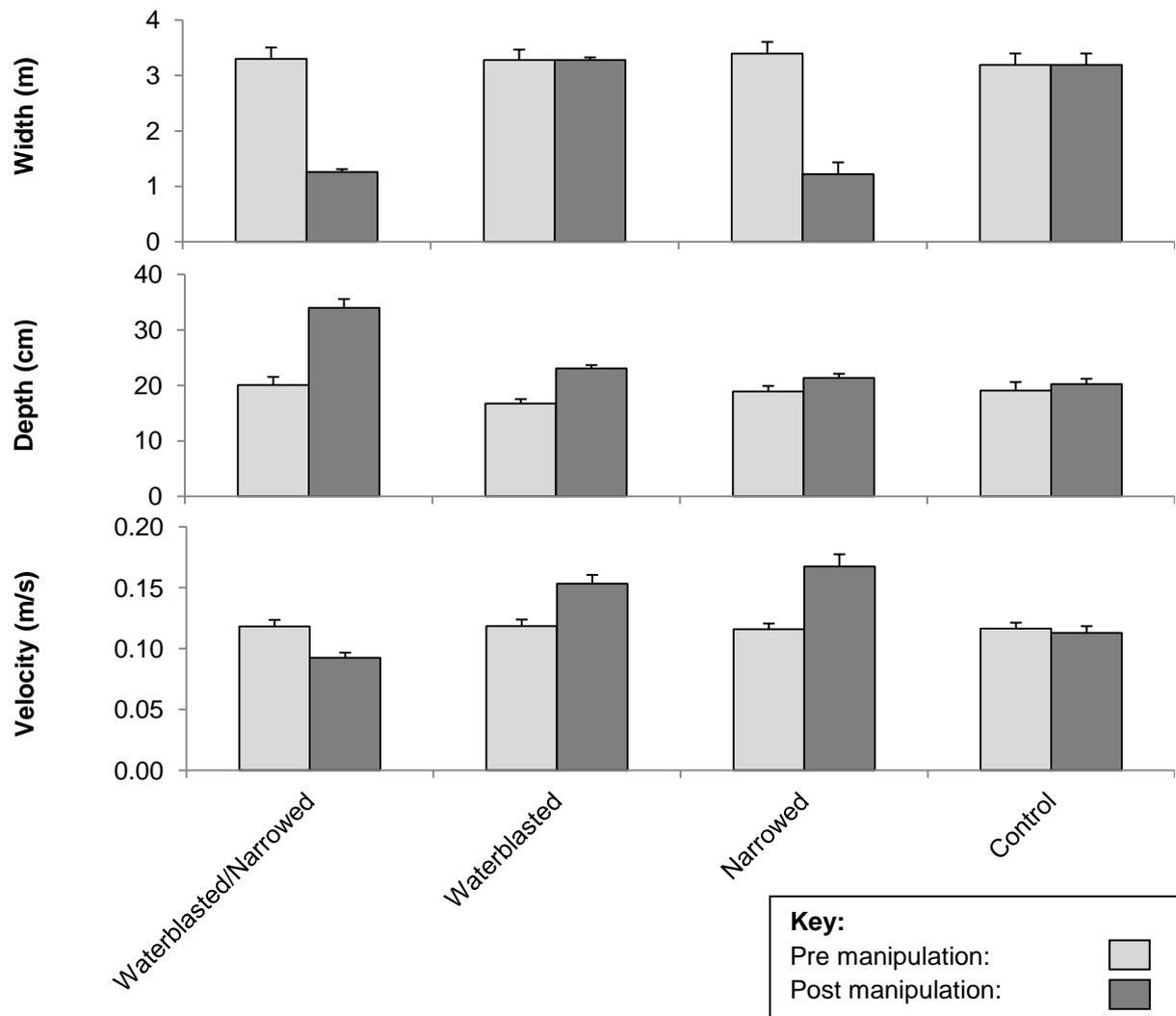


Figure 5. Average change (and standard error) pre- and post-treatment application, for habitat variables among treatment blocks in a two by two factorial design of water blasting to remove surface sediment and/or narrowing of the channel to increase velocity.

Narrowing was intended to increase velocity through the stream channel, however when narrowing was applied with water-blasting this created a deeper channel which in turn reduced velocity, and possibly reduced the effectiveness of this combined treatment type (Figure 5.). This suggests the use of one treatment type may be more beneficial to stream recovery than combining treatments due to the three-way interaction involving water-blasting, narrowing and time that affected water depth and velocity (Table 2.). The control, with no treatments applied gradually increased in sediment depth over time, suggesting that if nothing was done about the issue of sedimentation, sediment depth would probably increase (Figure 4.).

The NMDS community composition ordination broadly separated the sediment removal invertebrate communities into non-water-blasted (high axis 1 and 2 scores, with high sediment) and water-blasted groups (low axis 1 and 2 scores, low sediment), whereby sensitive and tolerant species are associated with water-blasted and non-water-blasted areas respectively. However, sediment removal by water-blasting was more effective at certain blocks (1, 2 and 5) and less effective in others with deeper sediment (3 and 4), resulting in some water-blasted sites grouping with non-water-blasted sites on the ordination (Figure 6.).

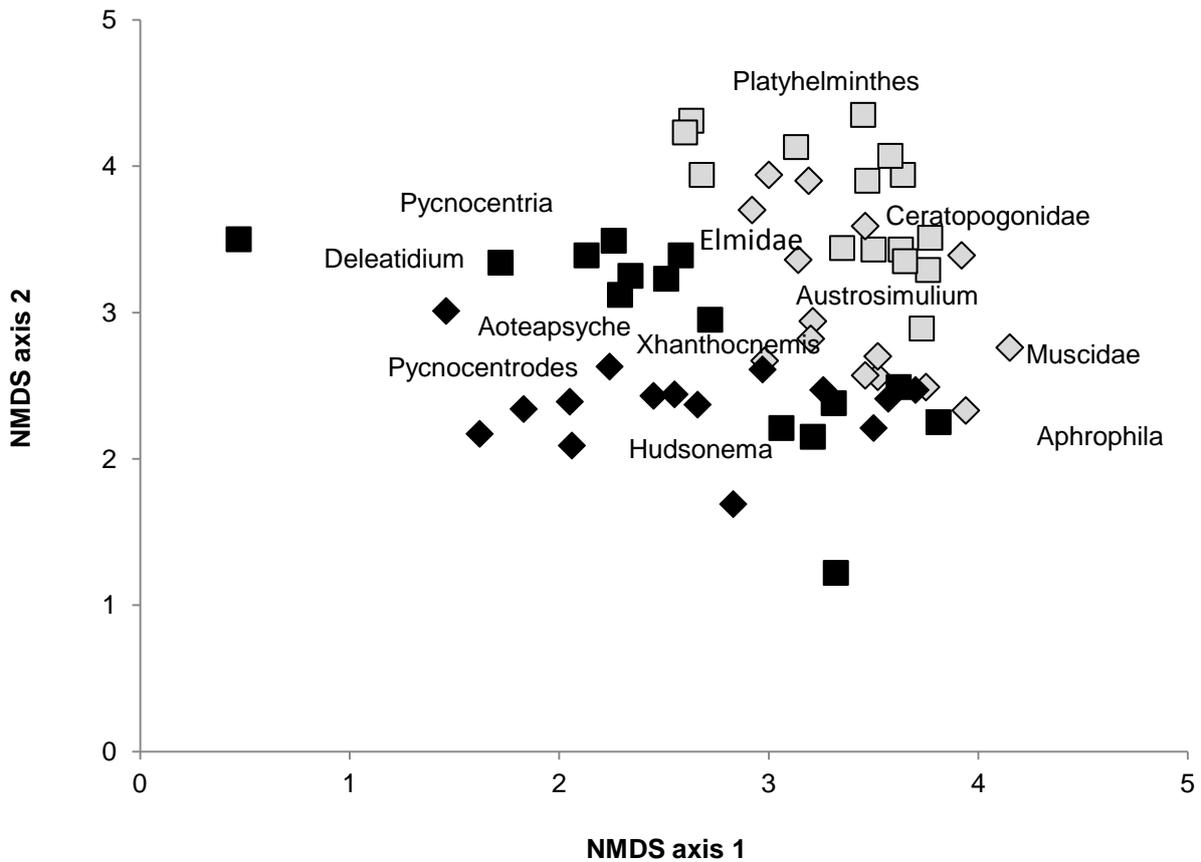


Figure 6. Community (symbols; stress, 0.14) and species (taxa names; stress, 0.13) NMDS ordination on invertebrate abundance, where abundances were square-root transformed and under a Bray Curtis dissimilarity matrix. Treatments were setup as a two by two factorial design of water blasting (to remove surface sediment) and/or narrowing of the channel (to increase velocity).

Key:		Narrowed:	
Water blasted:		Yes	No
Yes	◆	■	■
No	◇	□	□

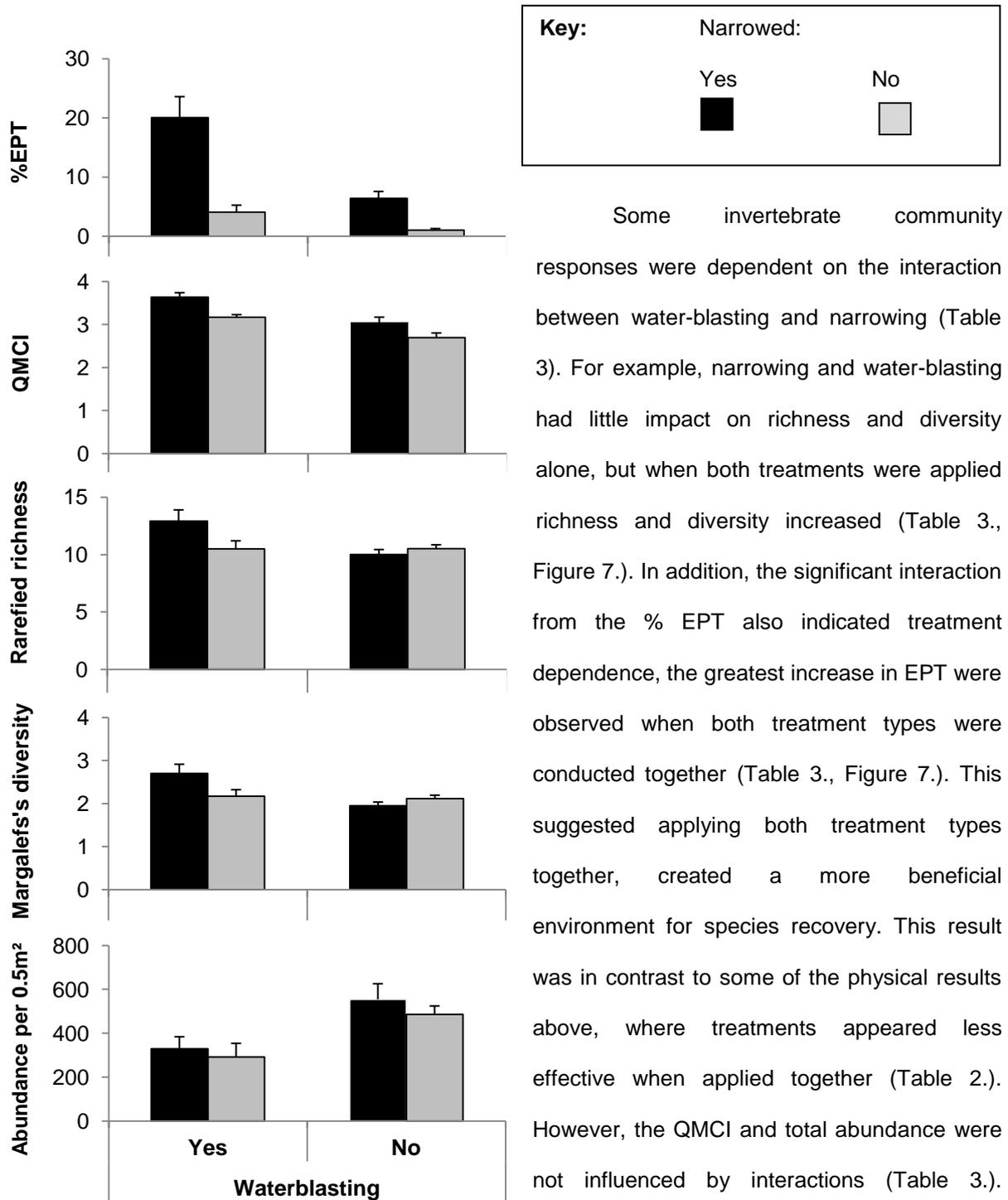


Figure 7. Average differences of the sediment experiment community responses (and standard error) of percentage of EPT taxa, QMCI, rarefied richness, Margalef's diversity and abundance (per 0.5 m²) among treatment blocks in a two by two factorial design of water blasting to remove surface sediment and/or narrowing of the channel to increase velocity.

Some invertebrate community responses were dependent on the interaction between water-blasting and narrowing (Table 3). For example, narrowing and water-blasting had little impact on richness and diversity alone, but when both treatments were applied richness and diversity increased (Table 3., Figure 7.). In addition, the significant interaction from the % EPT also indicated treatment dependence, the greatest increase in EPT were observed when both treatment types were conducted together (Table 3., Figure 7.). This suggested applying both treatment types together, created a more beneficial environment for species recovery. This result was in contrast to some of the physical results above, where treatments appeared less effective when applied together (Table 2.). However, the QMCI and total abundance were not influenced by interactions (Table 3.). Abundance of organisms was significantly affected by water-blasting, whereby abundance declined after this treatment was applied (Table 3.). This reduction in abundance in water-blasted sites may be an indication that species have not yet fully recolonised this habitat at the time of sampling. However, we can see that the

species that have recolonised to the water-blasted sections are more sensitive, supported by the greater % EPT scores in both water-blasting treatments in comparison to non-water-blasted treatments (Figure 7.). The treatment effects on QMCI were independent, indicating although the presence of narrowing had positive effects, water-blasting showed greater increases in sensitive species, also supported by the separation of water-blasted and non-water-blasted sites in the community ordination (Table 3., Figure 6 and 7.).

Table 3. Differences in community responses of percentage of EPT taxa, QMCI, rarefied richness, Margalef's diversity and abundance (per/0.5 m²) per treatment, analyses by 2-way ANOVA where proportions were arcsine-square-root-transformed and abundance were log-transformed among treatment blocks in a two by two factorial design of water-blasting to remove surface sediment and/or narrowing of the channel to increase velocity. *F ratio for the MAOVA is an approximation calculated by R. The Pillai value for narrowed, water-blasted and the interaction are 0.46 0.51 and 0.18 respectively.

		df	MS	F*	P
MANOVA	Narrowed	1	0.46	8.99	<0.01
	Water blasted	1	0.51	11.01	<0.01
	Interaction	1	0.18	2.27	0.06
	Residuals	56			
% EPT	Narrowed	1	1706.46	29.78	<0.01
	Water blasted	1	1048.81	18.30	<0.01
	Interaction	1	421.75	7.36	<0.01
	Residuals	56	57.30		
QMCI	Narrowed	1	2.50	16.19	<0.01
	Water blasted	1	4.33	28.04	<0.01
	Interaction	1	0.05	0.35	0.55
	Residuals	56	0.15		
Rarefied richness	Narrowed	1	13.51	2.09	0.15
	Water blasted	1	30.70	4.74	0.03
	Interaction	1	32.05	4.95	0.03
	Residuals	56	6.47		
Margalef's diversity	Narrowed	1	0.49	1.58	0.21
	Water blasted	1	2.46	8.01	0.01
	Interaction	1	1.85	6.04	0.02
	Residuals	56	0.31		
Abundance	Narrow	1	662550	0.85	0.36
	Water blasted	1	43041	13.06	<0.01
	Interaction	1	3542	0.07	0.79
	Residuals	56	50728		

DISCUSSION

Excessive sedimentation has pushed many streams into hysteresis, whereby current restoration techniques are not effective at improving communities or in-stream habitat (Jansson *et al.* 2007, Lake *et al.* 2007, Miller *et al.* 2010). Many restoration projects do little in the way of encouraging sediment removal but focus on reducing further inputs (Lowrance *et al.* 1984, Dillaha *et al.* 1989, Craig *et al.* 2008), assuming streams possess the natural capacity necessary to remove the sediment build-up. Therefore, in many situations, as seen in Canterbury Plains, in-stream damage may be beyond the influence of these restoration techniques, and removing the potential threats does not typically result in fine sediment reductions or positive community changes (Greenwood *et al.* 2012). By identifying factors limiting invertebrate communities and the role of sediment in particular, my results indicate ways of dealing with high levels of sediment in streams requiring restoration.

The rock-basket experiment results suggest high quantities of fine sediment are the most important factor restricting recovery of sensitive species at Riversdale Stream. The addition of sediment-free habitat resulted in measurable positive community shifts, providing evidence for sensitive species preferring and moving into low sediment habitats. In addition, when providing this alternative habitat for invertebrate communities, I found greater variability in community composition, in comparison to communities in natural original habitat. This was possibly due to the greater variability in habitat provided, in contrast to the original homogenous and degraded environment. Moreover, the greater dissimilarity among communities in the improved habitat observed in my study suggests modification of one aspect of habitat quality might result in a continuum of habitat diversity when coupled with naturally occurring habitat features. This is consistent with the nested habitats hypothesis (Hylander *et al.* 2005), where the sediment gradient reflects changing heterogeneity and complexity with sediment-free habitats having a wider range of microhabitats, and thus a wider range of community types can occupy them in comparison to sediment-impacted locations.

Previous studies show that most invertebrate species prefer sediment-free conditions, and even small additions of sediment can deleteriously affect richness, abundance, and composition of invertebrates (Rabeni *et al.* 2005, Downes *et al.* 2006, Connolly and Pearson 2007, Larsen *et al.* 2011). However, when sediment-free habitat was provided in an already sedimented stream, I observed species recovery and positive community changes. Additionally, these newly formed communities associated with the added habitat contained more sensitive taxa than the natural habitat,

indicated by greater proportions of EPT and other invertebrates that score highly on the MCI scale. This discovery is a positive first step in mitigating the effects of deposited sediment and indicates that sediment removal should positively influence invertebrate communities to reflect a less degraded environment. This result is particularly important when control sections in the sediment removal experiment resulted in a further build-up of sediment despite livestock exclusion, indicating issues around sediment may only worsen. This improvement in community structure, also observed in other studies (Owens *et al.* 2005, Larsen *et al.* 2011), implies the issue of heavy sedimentation may only worsen without active interference and underlines the importance of developing techniques for sediment removal.

The sediment removal experimental results suggested that active sediment removal using a combination of increasing stream velocity and actively flushing sediment from the streambed could be successful at removing deposited fine sediment from the streambed. Subsequently, sediment removal may be an option for stream restoration and when used in combination with techniques that address the causes of degradation, should be most effective at permanently removing deposited sediment and restricting future sediment deposition. The rate at which sediment was removed depended on the treatment applied, which in turn was strongly dependent on the amount of surrounding sediment remaining to be moved. The reduced width and increased velocity of the narrowed treatments meant sediment was removed at least twice as fast as water-blasted only treatments (post-water-blasting). However, sediment levels at the end of this experiment in narrowed treatments had more sediment left to remove than water-blasted sites. Thus, narrowing the stream channel was effective at removing sediment in the short-term but generally slower than water-blasting at removing sediment.

Sediment removal via channel narrowing is a continual process, and may take a considerable time to reduce sediment build-up. For example, a section of this stream with 10 cm of sediment cover is likely to take over two years to clear sediment, assuming the rate observed is constant throughout this time. Furthermore, it is not known if narrowing flow will result in the complete removal of sediment when large substrates are heavily embedded and armoured into the fine sediment. Additionally, if channel depth increases due to the removal of sediment, as seen in the water-blasting treatment, this may slow velocity and reduce the rate of sediment removal (Knapp and Matthews 1996, Goodwin *et al.* 1997). Moreover, sediment removal in this study was observed over the spring and summer period where flows were often low, larger and more flushing flows would be expected during winter,

suggesting sediment removal methods and sediment flushing through natural flows may be much more successful when these larger and more flushing flows occur. Recent observations of Riversdale Stream exhibit this sort of response from a high-flow event where areas of the stream with largely intact banks appear to have had substantial sediment flushing, but the specific effects have not yet been measured.

Further and longer-term studies must be done to determine the full potential of channel narrowing as a sediment removal technique. However, if a stream has become over-widened due to human induced causes, then narrowing the channel and bank repair should be a priority for restoration regardless of issues of in-stream sediment. Moreover, the addition of large stable substrate such as, boulders along fragile and already eroded stream banks may allow for the protection of these areas as well as increased water velocity through the middle of the channel that may remove deposited sediment (Lefrançois *et al.* 2007). The addition of boulders to bank sides in the narrowing treatment also created reduced velocity between boulders and therefore may allow the controlled build-up of sediment between these added substrates aiding the narrowing process. Once livestock are fully removed riparian vegetation could eventually use this sediment build-up and incorporate it into the bank sides to produce a permanently narrowed stream (Lyons *et al.* 2000, Carline and Walsh 2007).

In comparison to channel narrowing, water-blasting quickly and completely removed surface sediment to substantial depths in most cases, resulting in the remaining substrate being larger and more loosely packed. Stream communities might therefore be expected to recolonise immediately after sediment removal, resulting in a much faster recovery response. Nevertheless, my results indicate that this may not have been the case as abundances of total invertebrates in the water-blasted sections were significantly lower than non-water-blasted sections. However, this reduced abundance could result from a reduction of pollution-tolerant species in the water-blasted sections, such as oligochaets that generally occur in greater densities than more sensitive species (Stone *et al.* 2005). Restoration techniques can result in declines in invertebrate abundances due to shifts in community dominance (Albertson *et al.* 2011). Additionally, other studies show increased abundance of invertebrates due to natural sediment movement off the streambed (e.g. Carline and Walsh, 2007). However, the processes that led to this sediment removal were unlikely to have completely removed previous communities and therefore these abundances are likely to include sediment tolerant species.

In addition, water-blasting also resulted in a large pulse of sediment being washed downstream and a much larger disturbance to stream communities over the water-blasted area. Already a restoration tool is being developed to remove in-stream sediment without this problem. The Sandwand is designed to minimise the sediment pulse in receiving communities by pumping deposited fine sediment out of the stream (Strum and Barth 2012). This tool shows promising potential but, little is known about its long-term success, or its effect in highly sedimented streams.

In this study, there was no evidence of fine sediment re-depositing on the streambed in the manipulated treatments after five months of observations. This may be due to the increases in velocity and turbulence over the newly uncovered and diverse substrates in these treatments, in addition to the faster velocities created during narrowing. However, this may not be the case in the long-term or during extended periods of low flows especially with livestock disturbance, and therefore water-blasting may require repeated applications. This may be particularly important if sediment sources are not removed, as water-blasting alone does not address the causes of sedimentation. However, combining water-blasting and narrowing should mean fine sediments do not have the opportunity to resettle within the narrowed stream channel with faster velocity (Lefrançois *et al.* 2007), therefore, providing permanent high quality habitat and allowing recovery of sensitive species.

The most important result was that removal of fine sediment from the stream that was causing the hysteresis effect resulted in the recovery of sensitive invertebrates. The removal of sediment also saw positive invertebrate community changes similar to those observed in the smaller-scale rock-basket experiment. However, there was diversion of invertebrate communities from non-water-blasted (high sediment) and water-blasted treatments (low sediment). This separation was due to sensitive and tolerant species associating with water-blasted and non-water-blasted areas, respectively. For instance, *Deleatidium* mayflies and *Pycnocentria* Caddisflies, sensitive EPT taxa, were only found in water-blasted treatments with low sediment and contributed greatly to the increases in % EPT and QMCI scores in these areas.

The combination of mechanical and hydrological movement of sediment was also the most effective at returning sensitive species indicated by the significant interactions affecting % EPT, diversity and richness. Thus, the combination of water-blasting and narrowing resulted in a greater change towards a pollution-sensitive invertebrate community than either techniques alone. This occurred despite the collective effect of these treatments resulting in a slower velocity due to a

narrowing/deepening of the channel. EPT taxa are generally associated with faster water velocities and larger substrates in Canterbury (Greenwood *et al.* 2012), so the improvements in invertebrate communities were somewhat unexpected given the slower velocities of the combined treatments. However, the combination of these methods resulted in significantly lower post-manipulation sediment depth than other treatments. This alone could have had important effects on the invertebrate communities response and again suggests that these changes seen in these communities is closely dependent on the success of sediment removal. However, previous studies suggest that EPT are also associated with deeper and narrower channels (e.g. Parkyn *et al.* 2003 and Greenwood *et al.* 2012), which may also have had an influence on their movement and habitat choices in this experiment.

Invertebrates are an important tool for identifying habitat derived changes at local scales (Armitage *et al.* 1987). Restoration success is often determined via observing changes in these communities, however, success or improvements in community compositions are not readily found (Palmer *et al.* 2010). Therefore, it is important to discover what is limiting stream recovery and then actively address these issues. My major conclusion is that high sediment loads in streams restrict aquatic communities, and the removal of this sediment allows their recovery. The practical and logistical application of sediment removal needs further development.

Invertebrate community recovery is an important indicator of restoration success however, different restoration projects will often have other goals such as improving habitat for fish. As discussed in Chapter One, intermittent streams require good in-stream habitat to support both fish and invertebrate communities during drought periods and in particular pools can have great importance. It was observed in this study when Riversdale Stream began to dry that the water-blasted blocks remained as large pools for much longer than non-water-blasted sections. This is likely to have been caused by the deepening of the channel, although the exposure of different sized substrates and unclogging of indices may also have influenced this, suggesting the removal of fine sediments via water-blasting will also aid in the recovery of fish habitat.

In my study, the greatest improvements and changes in community state occurred in areas with the greatest degradation, the current populations available in this stream had the most to gain from these habitat improvements. For example, the rock-basket addition resulted in the greatest and most positive changes towards a sensitive species dominance in communities in the high sediment areas. However, some authors suggest these highly degraded areas should not have high priority for

conservation, due to the large effort required to fully restore historic conditions (Palmer and Filoso 2009). Nevertheless, streams such as this one, can display more obvious signs of recovery in comparison to less degraded habitat and therefore may be a great place to learn about restorations techniques. Further improvements of community state to areas with already good habitat quality may be limited by other factors that dampen their response, such as the need for external colonisation, oviposition habitat (described in Chapter 2), or habitat modifications insufficiently improving habitat for sensitive species (Palmer *et al.* 1997).

Additionally, it appears that current restoration techniques like riparian management are more effective on less degraded, largely intact streams and therefore, restoration priorities should be focused here (Roni *et al.* 2008, Greenwood *et al.* 2012). This makes sense when there are intact colonisation sources (Chapter Two). Ultimately, heavily degraded or polluted streams in headwaters such as Riversdale Stream are also likely to contribute greater pollution to downstream habitats and larger rivers unless their problems are dealt with. This therefore makes Riversdale Stream and other heavily degraded headwater streams a priority for restoration activities although restoration goals should be more conservative and realistic for the conditions. Additionally, the positive results of my study suggest further development of restoration techniques that focus on factors limiting stream restoration, such as sediment will have worthwhile outcomes, where otherwise, streams such as Riversdale could be seen as a lost cause.

CHAPTER FOUR

Management recommendations



Seeing the light, from Riversdale Stream

APPLICATION OF THE FINDINGS AT RIVERSDALE STREAM

Recently, restoration ecology has emerged as a very important branch of science because of the extent of environmental degradation. Restoration ecology attracts great public attention among interest groups and individuals, tangata whenua, landowners, and managers, emphasising the importance to have a clear understanding of what restoration is, and how it can be applied. Restoration activities can occur along a continuum over different levels of degradation, and therefore realistic and appropriate levels of restoration effort and targets should be used to aid ecosystem recovery. Therefore, I define restoration as an attempt to return a degraded ecosystem to a more healthy state. This does not necessarily need to be a historic state, however outcomes may be more successful when considering history and current land-use (Nilsson *et al.* 2005).

Streams and rivers are among the most impacted ecosystems in the world (Vörösmarty *et al.* 2010), and restoration has raised great optimism and pressure to return these to a better functioning condition (Parkyn *et al.* 2003). However, improvements in invertebrate communities, a commonly used indicator of environmental health, for various reasons are not often found, resulting in project failure (Palmer *et al.* 2010). This can be very discouraging for community groups, who often plan and manage projects themselves, and frustrating for ecologists when theory and practise do not coincide (Palmer and Filoso 2009).

Despite the broad range of restoration techniques, several common factors appear to limit the success of projects. Water quality and quantity, erosion and sedimentation, fragmentation, and loss of connectivity to species pools, prevent many projects from achieving their full potential (Wood and Armitage 1997, Winterbourn *et al.* 2007, Roni *et al.* 2008, Palmer *et al.* 2010, Didham *et al.* 2012). Due to a lack of understanding of surrounding broader-scale issues, each of these factors limiting invertebrate communities can result in restoration failure (Beechie and Bolton 1999, Roni *et al.* 2002, Wohl *et al.* 2005, Beechie *et al.* 2008). Restoration success can be improved by developing techniques that focus on the factor most limiting invertebrate recovery, and generally this involves enhancing invertebrate communities at a restoration site in two ways. Firstly, additional and pollution-sensitive species may be added to the site, this is most often achieved through natural colonisation. Secondly, positive changes to current communities can occur through changes in community dominance from pollution-tolerant to sensitive species without the need for external colonisation or stream connectivity.

In this chapter

In this chapter, I summarise key issues surrounding the factors limiting invertebrate community responses to stream restoration focusing on the importance of the “field of dreams” hypothesis and colonisation sources, and how these influence prioritisation and realistic goal setting. I will highlight relevant findings from my case study of Riversdale Stream, a waterway that has become highly degraded by agricultural influences, particularly livestock damage to stream banks. Additionally this stream is connected to a diverse species pool, which reduces complications and assumptions of restoration, and allows further advances in determining what commonly can limit restoration efforts. For this stream, the most important limiting factor is excessive sediment, and I make conclusions based on findings from Riversdale Stream that are widely applicable to other streams in similar situations.

How can we improve restoration success?

To begin with, there are many assumptions that surround stream and river restoration, many of which are not well tested (Wohl *et al.* 2005, Jähnig *et al.* 2011). One common underlying assumption is that restoration of stream morphology and habitat to a pre-degraded state will lead to the recovery of aquatic organisms (Violin *et al.* 2011). This assumption relies on improving habitat to allow for colonisation (Palmer *et al.* 1997). However, making this assumption does not always consider if there is a connection to a colonisation source. Although little is known about the specific time scale of community recovery during restoration, if connections to source populations are lost through landscape barriers, the time for recovery may be substantially longer (Lake *et al.* 2007). Examples of the kind of connectivity necessary for the faster recovery may be a nearby stream that contains desired species, the creation of dispersal corridors, or connections to intact habitat (Petersen *et al.* 2004). Without these kinds of connections, colonisation is likely to be limited and may result in few biodiversity improvements, despite efforts to improve habitat (Sundermann *et al.* 2011).

Riversdale Stream is surrounded by high quality habitat, allowing it to be connected to a regional species pool (Chapter One and Two). Similar connections in other studies have been associated with faster recovery of invertebrate populations post-restoration (Huxel and Hastings 1999). Connection to a diverse species pool should therefore, be reflected in restoration goals. By measuring the proximity of a restoration site to high quality habitat patches, and determining if that

habitat has additional species desired at a restoration site, the extent of connections to a local species pool can be determined. Furthermore, a stream's restoration potential is also important to consider. For example, Riversdale Stream's should not be expected to mimic that of a forested stream. If habitat restoration of this stream is successful, colonisation of forest-dwelling species, specifically species that may only occur in beech forests or species that prefer not to disperse across non-forested habitat is unlikely (Chapter Two). Habitat connectivity, such as dispersal corridors could increase the likelihood of colonisation, but consideration of appropriate target species, such as grassland species is more likely to ensure success (Figure 1.). Thus, goal setting and being realistic about what is possible is important.

External colonisation to a restoration site depends on the site being connected to a species pool. However, in many cases a connection is unlikely due to drastic changes in the landscape creating multiple barriers for colonisation (Parkyn *et al.* 2003). Nevertheless, this does not mean that an unconnected restoration site is automatically doomed to restoration failure or a well-connected site will automatically be a success. The connectivity of a site *must* be considered in restoration planning and goal setting for individual projects, as the connectivity and landscape context of a site will likely influence the speed and short- or mid-term success of biodiversity returning. Careful consideration of all the factors determining colonisation success will be important (Figure 1.)

Riversdale Stream's connectivity can be evaluated in two ways. Firstly, Riversdale is likely to be connected to a regional species pool because of the stream's close proximity to diverse assemblages of aquatic invertebrates. This kind of connectivity might not occur for streams in landscapes that are not adjacent to or contain intact habitat, such as in urban or agriculturally dominated areas. Secondly, while connection increases the likelihood of colonisation, Riversdale's connectivity may be hindered by what could be conceived as a hostile environment of open grassland habitat that surrounds it in comparison to forested and scrubland habitat for aquatic insects. Consequently, even streams with close connections to intact habitat cannot be assumed to be colonised if the habitat surrounding the restoration site is unsuitable (Figure 1.).

A restoration site that has strong potential for external colonisation may provide a more optimistic scenario for additional species to colonise the stream. However, diversity and composition of stream communities depend strongly on factors at multiple scales within catchments (Townsend *et al.* 2003, Townsend *et al.* 2004, Palmer *et al.* 2010). Factors to consider when selecting target species

should be: flight periods, connecting habitat suitability, dispersal ability, habitat requirements (of multiple life stages), and abundance of desired species (Figure 1.). These factors should be considered when deciding restoration goals and indicators of success. For example, the distribution of adult aquatic insects is strongly influenced by vegetation densities with important differences between dispersal through grassland, scrubland and forest habitats (Petersen *et al.* 2004, Winterbourn *et al.* 2007, Didham *et al.* 2012). Some aquatic insects are restricted to flight only in particular vegetation types (Winterbourn *et al.* 2007). This has important implications for dispersal to and colonisation of Riversdale Stream and will likely limit the return of specific species, for example forest specialists

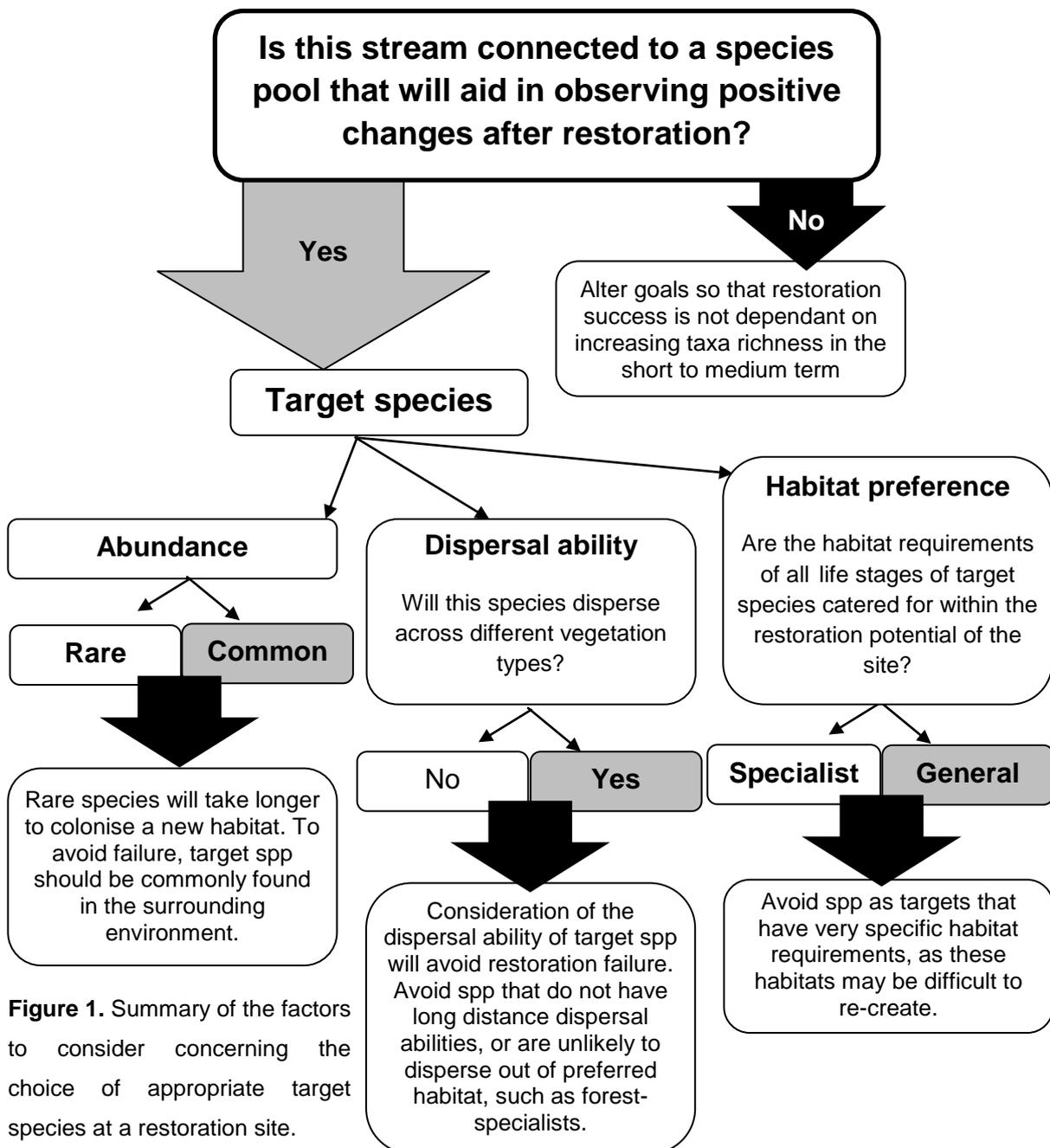


Figure 1. Summary of the factors to consider concerning the choice of appropriate target species at a restoration site.

(Chapter Two). This scenario (i.e. partial connectivity) likely a common restriction to restored streams, no matter on how well habitat restoration has improved in-stream conditions. Therefore, realistic goals on improving diversity and target species must be set to reduce disappointment.

Determining the limiting factors in stream restoration

Riparian management is universally used for protecting and improving stream habitat, often involving fencing to exclude livestock and vegetation addition to banks to create buffer zones, which absorb and trap potential pollutants, and add detritus (food resource), and shade the stream (Parkyn *et al.* 2003, Craig *et al.* 2008). However, invertebrate responses to this type restoration management have been variable and very context-dependant (Parkyn *et al.* 2003, Rhodes *et al.* 2007, Death and Collier 2010, Palmer *et al.* 2010). Variation in invertebrate responses is often attributed to the quantity and purpose of management and initial level of degradation, particularly to pollutants in the stream, such as fine sediments (e.g. Greenwood *et al.* 2012). While riparian buffer strips may aid in reducing further input of these pollutants, in the short term, riparian management is unlikely to have any great impact on already high sediment levels because many of these stream have been hydrologically altered and no longer have the capacity to flush sediments naturally. This may be the case for Riversdale Stream, particularly in areas that have become over-widened with extensive bank damage from livestock pugging. Therefore, additional restoration assistance will be required to facilitate sediment flushing in these areas, so that improvements in invertebrates can be seen.

Additionally, many other restoration projects focus solely on habitat enhancements and modifications and therefore, restoration of in-stream habitat is one of the more common river restoration practices (Purcell *et al.* 2002, Bernhardt *et al.* 2007). In streams with largely homogenised environments, habitat restoration is most commonly focused on habitat additions, such as introducing boulders and wood, or channel reconfiguration at the reach-scale (Miller *et al.* 2010). The assumption that increasing habitat quality will improve biodiversity is based on research demonstrating habitat heterogeneity and species richness are positively correlated (Brown 2003, Helfield *et al.* 2007, Kail *et al.* 2007). However, positive invertebrate responses post-restoration efforts are not often associated with this restoration method (Palmer *et al.* 1997). This is not to say that improving habitat structure is not important, effective restoration projects should recapture the habitat structure and particularly the complexity of reference or pre-degraded conditions. However, there is often a more pressing matter

restricting recovery in most documented examples (Suding *et al.* 2004, Palmer *et al.* 2005). This was observed during an oviposition habitat addition experiment at Riversdale Stream (Chapter Two) where oviposition habitat (boulders for hydrobiosid caddisflies) was added to the stream. In areas with deep sediment, there was little improvement in oviposition, indicating that while the addition of habitat may be an easy option and has the potential to have important influences for biodiversity recovery, it is not necessarily the factor most limiting invertebrate communities. Therefore, it is important to discover the most limiting factor and make that the focus of restoration efforts.

In the situation where the direct effect of increasing habitat diversity may be overwhelmed by other factors, an adaptive management approach may suit river restoration whereby the most limiting factor is addressed first (Roni *et al.* 2002, Roni *et al.* 2008). For example, a stream surrounded by plantation forestry may be most limited by sudden inputs of organic matter and sediment, post-felling (Quinn *et al.* 2004), an urban stream by water quality, heavy metals and other inorganic pollutants (Blakely and Harding 2005), and an agricultural stream by sediments and nutrients (Greenwood *et al.* 2012). In addition, they all might be limited by colonisation sources. Specific restoration techniques can be used to mitigate further damage of these land uses, however current techniques may not improve the damage already done.

Poor in-stream habitat is likely to be an important factor limiting the recovery of invertebrate communities, but teasing apart what aspect of this habitat is limiting communities may not be an easy task, and may require some experimentation involving a process of elimination. For instance, in Riversdale Stream the addition of small wire mesh baskets filled with stones above the fine sediment layer resulted in significant differences between natural and added habitat communities (Chapter Three). The communities that occupied the baskets contained on average much less sediment-tolerant species, indicating that sediment was limiting community recovery (Chapter Three). Moreover, the addition of boulders as oviposition habitat for certain species in Riversdale Stream led to more egg masses, and a lack of oviposition in the locally highly sedimented areas also indicated that excessive sediment limited this process (Chapter Two). Additionally, poor connectivity and poor water quality could be ruled out as the most limiting factor because of the location of Riversdale Stream (Chapter One). Finally, trialling methods of sediment removal also led to species recovery, with more sensitive taxa and improved community index scores in treated reaches, indicating more sensitive species in areas of stream that had significantly reduced sediment cover.

The sediment removal experiment in Riversdale Stream indicated that both sediment flushing and narrowing of the over-widened stream banks resulted in improvements in habitat quality due to reductions in fine sediment, indicating active sediment removal will aid in the recovery in invertebrate communities during restoration. Water-blasting resulted in fine sediment cover below the biodiversity limiting threshold of 20 % (Clapcott *et al.* 2011), and narrowing was effective at reducing sediment cover through increased stream velocity that encouraged sediment flushing. However, the combination of these techniques resulted in the most movement of sediment from the streambed; water-blasting quickly removed large amounts of sediment and narrowing of the channel ultimately targeted the underlying cause of sedimentation, which in this case was bank erosion and widening from cattle trampling.

Sediment removal techniques, however, need to be developed further, particularly to better define the balance between channel narrowing, depth, and increasing velocity to get the best results (Figure 2.). I observed interactions between physical channel alterations whereby depth and width, not surprisingly, had influences on the velocity and, therefore, affected the removal of deposited fine sediment (Chapter Three). Channel narrowing when combined with water-blasting may not require such extensive narrowing in future manipulations, due to increased velocity and turbulence over the newly exposed substrate, which hinders sediment resettling. Narrowing with natural sediment flushing through increasing velocity is likely to be most successful in flashy systems that receive periods of flooding or high flows, and therefore narrowing the channel will aid in natural sediment flushing (Figure 2.). By comparison, in more stable streams, like springs, sediment flushing alone may be more effective at removing deposited sediment, but the need for narrowing or bank protection will depend on the amount of bank damage, at individual sites (Figure 2.).

Overall, if heavy sediment deposits limit stream communities, improving the in-stream habitat by providing sediment-free environments should result in an increase in abundance of pollution sensitive species if a source population is available. Streams with altered channel structure and hydrology, may have lost the capacity to naturally flush sediment (Owens *et al.* 2005) and, therefore, will require active sediment removal techniques to improve habitat. This is indicated by little reduction in sediment or improvement in communities after a sediment source is removed (e.g. Greenwood *et al.* 2012). If improvements of invertebrate communities were not observed after sediment reduction, this would indicate a different limiting factor, such as colonisation-limitation. However, sedimentation

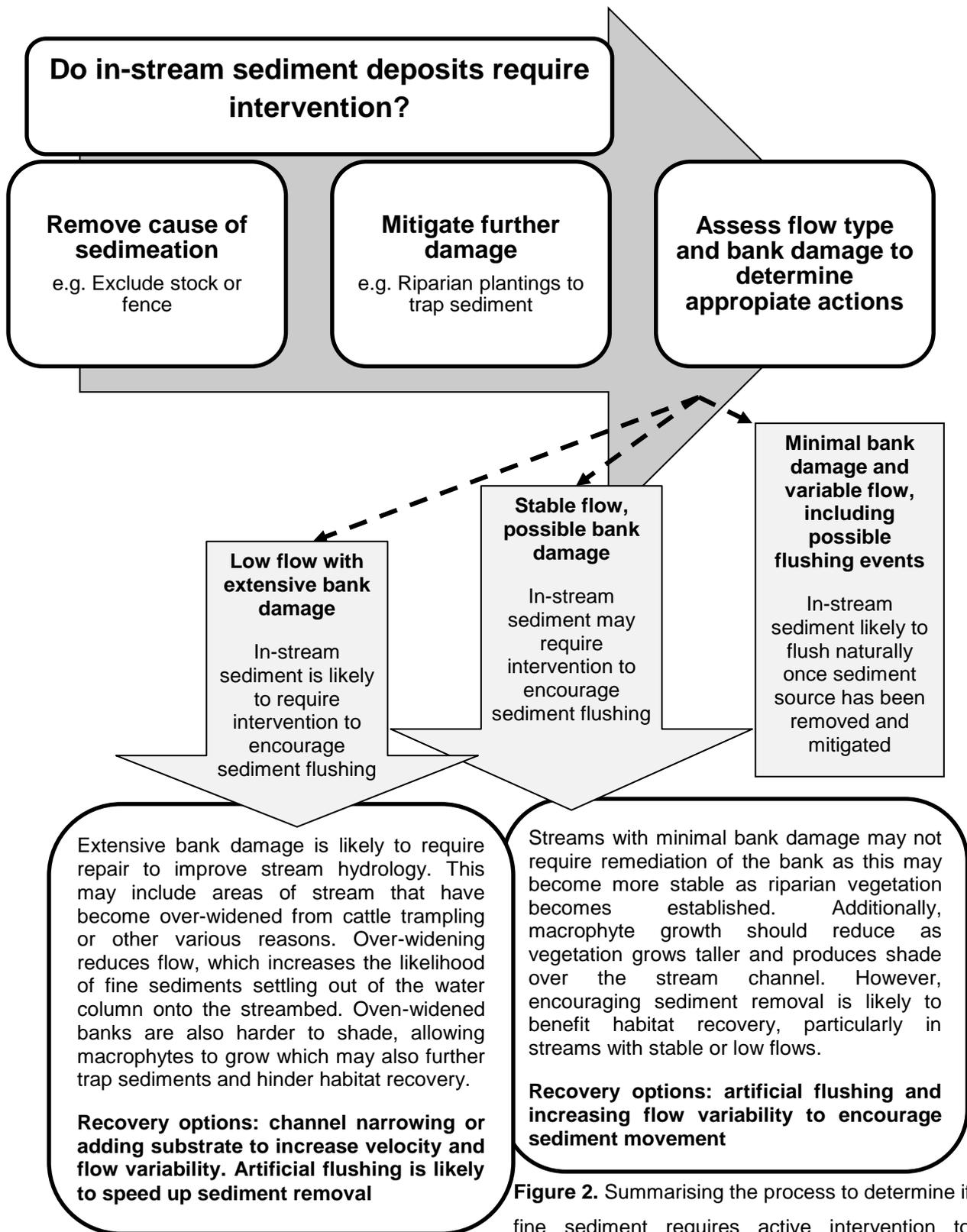


Figure 2. Summarising the process to determine if fine sediment requires active intervention to reduce deposits on the streambed.

is restricting invertebrate community recovery in Riversdale Stream and probably other streams in similar condition. Streams prone to receiving fine sediments should be a prioritised for restoration efforts, at least to reduce sediment entering the stream as this can pollute downstream waterways. Sedimentation can be minimised by fencing, excluding livestock and additions of bank vegetation, particularly grasses or other vegetation that creates bank stability and traps pollutants entering the stream. However, if levels of in-stream fine sediment are above 20 % total cover, improvements of invertebrate communities may be unlikely and restoration managers and goals should reflect this.

Priorities and goal setting for stream restoration

The degree of connectivity between habitat patches is likely to have important influences on the response time for restoration projects. Riversdale Stream has the potential to be connected to a regional species pool and generally, the greater the connectivity, the lesser the response time (Huxel and Hastings 1999). Restoration of streams as a result, might also be more effective when conducted at sites with minimal degradation, where the stressor can simply be removed or mitigated to allow for recovery of current hindered populations (Roni *et al.* 2008), and then this habitat may provide connectivity in the landscape (Figure 3.). Following this logic, areas coming outward from intact habitat also may be prioritised for restoration, allowing greater and more widely spread connectivity in the landscape and allowing in-stream habitat modifications to have a greater impact on biodiversity (Figure 3.).

Additionally, the accumulative effect of many small but nearby projects may also be taken into account to influence restoration success (Figure 3.). Working outward and around clusters of restoration projects, and especially towards intact habitat will create additional connectivity within the landscape. Streams from relatively intact landscapes are likely to require less restoration effort, though heavily degraded or polluted headwater-streams will likely contribute greater pollution to downstream habitats and recipient water bodies, possibly restricting downstream restoration efforts (Parkyn *et al.* 2003, Dodds and Oakes 2008). This therefore, also makes heavily degraded and polluted streams a priority for restoration activities, though goals may need to be more conservative and realistic for the conditions (Figure 3.).

The restoration potential and severity of degradation within a stream is very important to consider when goal setting, as it affects the target species and final habitat conditions. The habitat

degradation surrounding the stream may influence the dispersal ability of particular species. Riversdale Stream for instance, is surrounded by a landscape containing many different high-quality habitats that all contribute to the regional species pool. However, at the local landscape level Riversdale Stream is surrounded by grassland habitat and, therefore, the surrounding or connecting habitat from Riversdale to sources populations may still be unsuitable for dispersal of some species. Connectivity between habitats may be improved by creating dispersal corridors or connecting habitat patches. Alternatively, in the presence of a poor connection to a species pool, realistic restoration

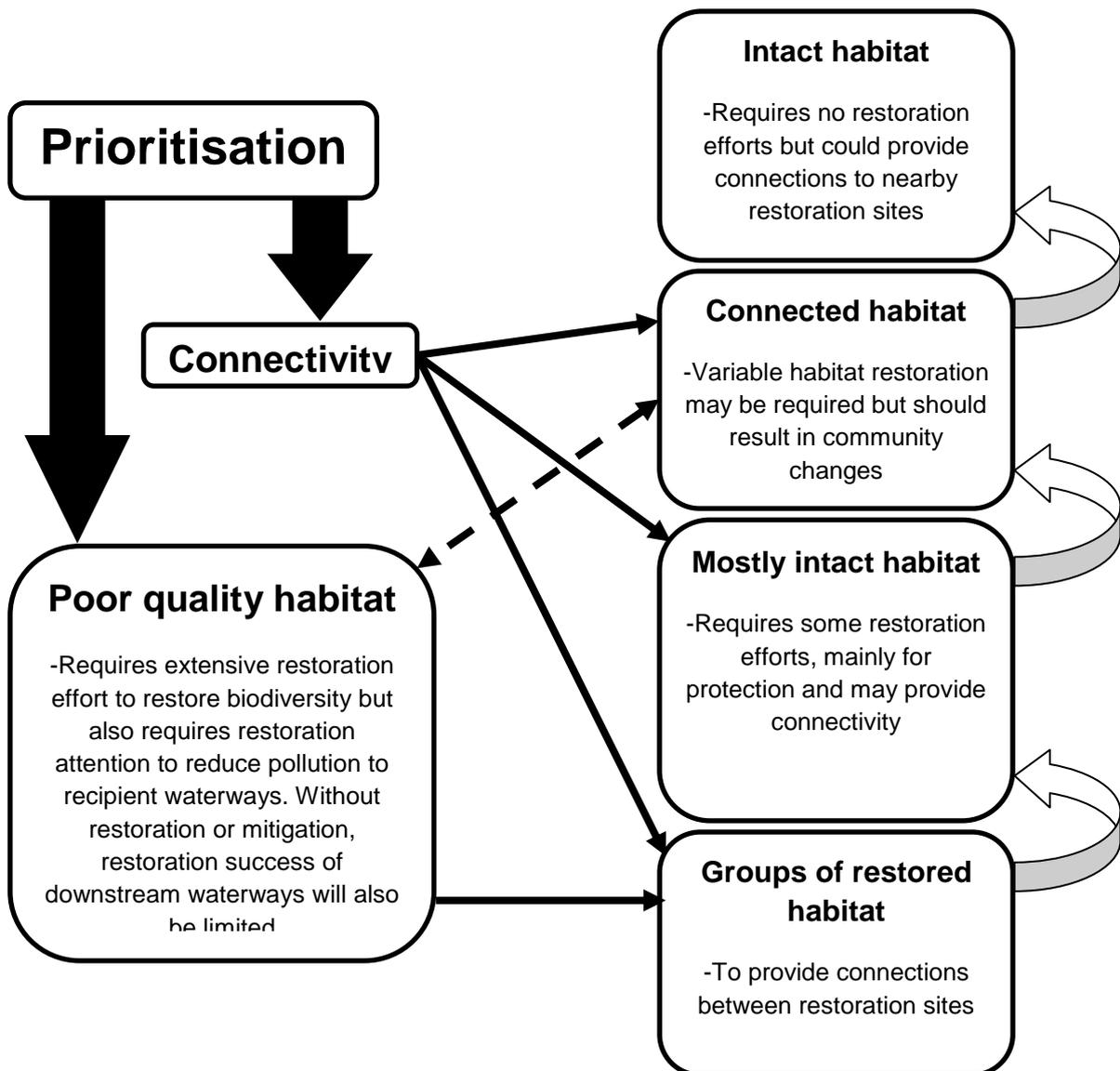


Figure 3. Summary of stream restoration prioritisation. Focus should be on streams with connectivity or creating connectivity, as these will likely show positive restoration outcomes and achieve goals. Additionally, Poor quality habitat and streams that increase downstream degradation should also be prioritised. However, these two different restoration priorities will need to have different goals as restoration outcomes are likely to be very different.

goals may focus on improving stream functioning, for example reducing nutrient loads in the water column that may also influence recipient water-bodies. Goals relative to improving invertebrate biodiversity in this case are more likely to succeed if focused on reducing community dominance of tolerant species, rather than relying on colonisation of new species.

Conclusions

Overall, many restoration projects are based on single or remote reaches of streams, where connectivity beyond the channel reach may not be considered (Palmer *et al.* 2005, Wohl *et al.* 2005). Therefore, restoration of streams with a potential connection, such as Riversdale Stream should produce more positive results. However, more commonly, many riparian planting projects are disconnected from one another and to intact habitat. Eventually this management will become more continuous, providing corridors of habitat suitable for dispersal enabling opportunities for colonisation from nearby populations (Petersen *et al.* 2004).

The majority of successful restoration projects suggest in-stream enhancement, when properly implemented, *can* potentially produce dramatic improvements in habitat and biodiversity (Roni *et al.* 2008). When enhancements are coupled with solutions to larger scale issues, local habitat improvement can lead to the long-term recovery of hydrology and water quality, sediment transport, and riparian conditions (Roni *et al.* 2008). Therefore, the most successful projects create large changes that reflect natural processes and focus on limiting factors (Roni and Quinn 2001, Roni *et al.* 2008). This was observed through positive invertebrate community change in Riversdale Stream when trials of in-stream enhancements (sediment removal) focused on factors that caused degradation (over widened banks, overcome by channel narrowing and increasing velocity). The challenge in restoration is to improve knowledge so that barriers preventing successful outcomes can be overcome.

Priorities can be set both among streams (discussed earlier) and within projects. Within projects, the priority must be focused towards whatever the greatest limiting factor is, ideally in an adaptive management approach. The limiting factor will change through the progression of the restoration project, and full restoration may only be complete when there is no longer anything limiting recovery. For example at Riversdale Stream, initially the most challenging limitation will be the amount of fine sediments dominating the substrate. However, once this factor is mitigated to acceptable levels, a lack of riparian vegetation (and associated benefits of) will likely limit Riversdale's invertebrate

recovery and from then perhaps it will be substrate heterogeneity and so on, until the site has reached its full potential. Restoration techniques that address the most limiting factor need to be prioritised so that restoration can achieve full potential within individual project constraints. Setting realistic and appropriate goals for restoration is likely to be a large deterrent of success, however it is also unrealistic to assume to have a generic set of recommendations for this and for restoration in general (Miller and Hobbs 2007).

ACKNOWLEDGEMENTS

I am going to keep this short and sweet, because words can only begin to describe the gratitude I feel for all those that have helped me on this journey. First and foremost, I need to thank Angus McIntosh, Angus thank you for encouraging and having faith in me to do this, your teaching and enthusiasm is incredibly inspiring. To my supervisors, Professor Angus McIntosh and Professor David Norton, and many other fantastic lecturers at Canterbury University your passion for your work has shaped my interest in ecology and made it possible for me to come this far. A huge thank you to the Mackenzie Charitable Foundation for funding this research, and for my scholarship, without this help my work would not have been easy. To Richard Smith from Mount White Station thanks again for granting me access to your land, and sorry once more for being a master's student and not from MasterChef.

I found myself incredibly lucky over the past few years to be surrounded by such amazing and talented ecologists. Thanks to Jon Harding, for co-leading FERG and putting a humourous spin on everything. Hamish Greig, Phil Jellymen and Jon O'Brien and of course Linda Morris and Milen Marinov for friendly advice and support. To everyone in the office, for making day-to-day life more interesting and enjoyable, and putting up with my 'occasional' grunts and sighs of frustration. I would also like to thank Jane Gosden, Helen Warburton, Mark Galatowitsch and various other part time residents at the Cass field station for all the entertainment over the summers. A special thank you to Amanda Klemmer, who as I write this is reading over the 95 pages of this thesis.

To all my wonderful friends, thank you firstly for putting up with me particularly over these last few months and secondly and predominantly to all those who I conned in to helping me with fieldwork on various occasions. Especially, Bonnie, Eric, Graham, Nate, Richard A and Richard R, (alphabetical order), who came with me on one particular weekend involving moving lots of rocks. To Emma, Samara and Dad for helping me proofread. Thank you to my family for all your words of support, encouragement and 'genuine' interest and of course, an extra special thank you to Eric Freme, I am not sure where I would be without you. Finally, to you all, I promise I will not make any immediate plans to do any more study in the near future and YES! The next round is on me (well, once I get a job that is).



REFERENCES

- Aerts R., Honnay O. 2011. Forest restoration, biodiversity and ecosystem functioning. *BioMed Central Ecology* **11**: 1-29.
- Albertson L. K., Cardinale B. J., Zeug S. C., Harrison L. R., Lenihan H. S., Wydzga M. A. 2011. Impacts of channel reconstruction on invertebrate assemblages in a restored river. *Restoration Ecology* **19**: 627-638.
- Alexander G. G., Allan J. D. 2007. Ecological Success in Stream Restoration: Case Studies from the Midwestern United States. *Environmental Management* **40**: 245–255.
- Allan J. D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology Evolution and Systematics* **35**: 257-284.
- Allan J. D., Johnson L. B. 1997. Catchment-scale analysis of aquatic ecosystems. *Freshwater Biology* **37**: 107-111.
- Angradi T. R. 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: A field experiment with biomonitoring applications. *Journal of the North American Benthological Society* **18**: 49-66.
- Armitage P. D., Gunn R. J. M., Furse M. T., Wright J. F., Moss D. 1987. The use of prediction to assess macroinvertebrate response to river regulation. *Hydrobiologia* **144**: 25-32.
- Arthington A. H., Balcombe S. R., Wilson G. A., Thoms M. C., Marshall J. 2005. Spatial and temporal variation in fish-assemblage structure in isolated waterholes during the 2001 dry season of an arid-zone floodplain river, Cooper Creek, Australia. *Marine and Freshwater Research* **56**: 25-35.
- Beechie T., Bolton S. 1999. An approach to restoring salmonid habitat-forming processes in Pacific Northwest watersheds. *Fisheries* **24**: 6-15.
- Beechie T., Pess G., Roni P. 2008. Setting river restoration priorities: A review of approaches and a general protocol for identifying and prioritizing actions. *North American Journal of Fisheries Management* **28**: 891-905.
- Bernhardt E. S., Palmer M. A. 2007. Restoring streams in an urbanizing world. *Freshwater Biology* **52**: 738-751.
- Bernhardt E. S., Sudduth E. B., Palmer M. A., Allan J. D., Meyer J. L., Alexander G., Follstad-Shah J., Hassett B., Jenkinson R., Lave R., Rumps J., Pagano L. 2007. Restoring rivers one reach at a time: Results from a survey of U.S. river restoration practitioners. *Restoration Ecology* **15**: 482-493.
- Bernhardt E. S., Palmer M. A., Allan J. D., Alexander G., Barnas K., Brooks S., Carr J., Clayton S., Dahm C., Follstad-Shah J., Galat D., Gloss S., Goodwin P., Hart D., Hassett B., Jenkinson R., Katz S., Kondolf G. M., Lake P. S., Lave R., Meyer J. L., O'Donnell T. K., Pagano L., Powell B., Sudduth O. 2005. Synthesizing U.S. river restoration efforts. *Science* **308**: 636-637.
- Blakely T. J., Harding J. S. 2005. Longitudinal patterns in benthic communities in an urban stream under restoration. *New Zealand Journal of Marine and Freshwater Research* **39**: 17-28.

- Blakely T. J., Harding J. S., McIntosh A. R., Winterbourn M. J. 2006. Barriers to the recovery of aquatic insect communities in urban streams. *Freshwater Biology* **51**: 1634–1645.
- Bo T., Fenoglio S., Malacarne G., Pessino M., Sgariboldi F. 2007. Effects of clogging on stream macroinvertebrates: An experimental approach. *Limnologica* **37**: 186-192.
- Bohn B. A., Kershner J. L. 2002. Establishing aquatic restoration priorities using a watershed approach. *Journal of Environmental Management* **64**: 355-363.
- Bohonak A. J., Jenkins D. G. 2003. Ecological and evolutionary significance of dispersal by freshwater invertebrates. *Ecology Letters* **6**: 783-796.
- Bond N. R., Downes B. J. 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., Lake P. S. 2003. Local habitat restoration in streams: Constraints on the effectiveness of restoration for stream biota. *Ecological Management and Restoration* **4**: 193-198.
- Bond N. R., Lake P. S. 2005. Ecological restoration and large-scale ecological disturbance: The effects of drought on the response by fish to a habitat restoration experiment. *Restoration Ecology* **13**: 39-48.
- Bond N. R., Lake P. S., Arthington A. H. 2008. The impacts of drought on freshwater ecosystems: An Australian perspective. *Hydrobiologia* **600**: 3-16.
- Boulton A. J., Hancock P. J. 2006. Rivers as groundwater-dependent ecosystems: A review of degrees of dependency, riverine processes and management implications. *Australian Journal of Botany* **54**: 133-144.
- Braccia A., Voshell Jr J. R. 2007. Benthic macroinvertebrate responses to increasing levels of cattle grazing in blue ridge mountain streams, Virginia, USA. *Environmental Monitoring and Assessment* **131**: 185-200.
- Brederveld R. J., Jähnig S. C., Lorenz A. W., Brunzel S., Soons M. B. 2011. Dispersal as a limiting factor in the colonization of restored mountain streams by plants and macroinvertebrates. *Journal of Applied Ecology* **48**: 1241-1250.
- Briers R. A., Cariss H. M., Gee J. H. R. 2002. Dispersal of adult stoneflies (Plecoptera) from upland streams draining catchments with contrasting land-use. *Archiv Fur Hydrobiologie* **155**: 627-644.
- Brooks S. S., Palmer M. A., Cardinale B. J., Swan C. M., Ribblett S. 2002. Assessing stream ecosystem rehabilitation: Limitations of community structure data. *Restoration Ecology* **10**: 156–168.
- Brown B. L. 2003. Spatial heterogeneity reduces temporal variability in stream insect communities. *Ecology Letters* **6**: 316-325.
- Bruns D. A. 2005. Macroinvertebrate response to land cover, habitat, and water chemistry in a mining-impacted river ecosystem: A GIS watershed analysis. *Aquatic Sciences* **67**: 403-423.
- Bunn S. E., Hughes J. M. 1997. Dispersal and recruitment in streams: evidence from genetic studies. *Journal of the North American Benthological Society* **16**: 338-346.

- Bunn S. E., Davies P. M., Kellaway D. M., Prosser I. P. 1998. Influence of invasive macrophytes on channel morphology and hydrology in an open tropical lowland stream, and potential control by riparian shading. *Freshwater Biology* **39**: 171-178.
- Bunn S. E., Thoms M. C., Hamilton S. K., Capon S. J. 2006. Flow variability in dryland rivers: Boom, bust and the bits in between. *River Research and Applications* **22**: 179-186.
- Burrows C. J. 1977. *Cass, history and science in the Cass district, Canterbury, New Zealand*. Christchurch, New Zealand: Department of Botany, University of Canterbury.
- Burrows C. J., Lord J. M. 1993. Recent colonisation by *Nothofagus fusca* at Cass, Canterbury. *New Zealand Journal of Botany* **31**: 139-146.
- Carline R. F., Walsh M. C. 2007. Responses to riparian restoration in the Spring Creek watershed, Central Pennsylvania. *Restoration Ecology* **15**: 731-742.
- Cipollini K. A., Maruyama A. L., Zimmerman C. L. 2005. Planning for restoration: A decision analysis approach to prioritization. *Restoration Ecology* **13**: 460-470.
- Clapcott J. E., Young R. G., Harding J. S., Matthaei C. D., Quinn J. M., Death R. G. 2011. *Sediment Assessment Methods: Protocols and guidelines for assessing the effects of deposited fine sediment on in-stream values*. Nelson, New Zealand: Cawthron Institute
- Clarke K. R., Gorley R. N. 2000. *PRIMER (Plymouth Routines in Multivariate Ecological Research) v5: User Manual/Tutorial*. Plymouth: PRIMER-E.
- Collier K. J., Smith B. J. 1997. Dispersal of adult caddisflies (Trichoptera) into forests alongside three New Zealand streams. *Hydrobiologia* **361**: 53-65.
- Collier K. J., Smith B. J., Baillie B. R. 1997. Summer light-trap catches of adult Trichoptera in hill-country catchments of contrasting land use, Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **31**: 623-634.
- Collier K. J., Wilcock R. J., Meredith A. S. 1998. Influence of substrate type and physico-chemical conditions on macroinvertebrate faunas and biotic indices of some lowland Waikato, New Zealand, streams. *New Zealand Journal of Marine and Freshwater Research* **32**: 1-19.
- Collier K. J., Croker G. F., Hickey C. W., Quinn J. M., Smith B. S. 1995. Effects of hydraulic conditions and larval size on the microdistribution of Hydrobiosidae (Trichoptera) in two New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* **29**: 439-451.
- Connolly N. M., Pearson R. G. 2007. The effect of fine sedimentation on tropical stream macroinvertebrate assemblages: A comparison using flow-through artificial stream channels and recirculating mesocosms. *Hydrobiologia* **592**: 423-438.
- Cowley D. R. 1978. Studies on the larvae of New Zealand Trichoptera. *New Zealand Journal of Zoology* **5**: 639-750.
- Craig L. S., Palmer M. A., Richardson D. C., Filoso S., Bernhardt E. S., Bledsoe B. P., Doyle M. W., Groffman P. M., Hassett B. A., Kaushal S. S., Mayer P. M., Smith S. M., Wilcock P. R. 2008. Stream restoration strategies for reducing river nitrogen loads. *Frontiers in Ecology and the Environment* **6**: 529-538.
- Crawley M. J. 2007. *The R book*. Chichester, UK: Wiley.

- Dance K. W., Hynes H. B. N. 1980. Some effects of agricultural land use on stream insect communities. *Environmental Pollution* **22**: 19-28.
- Davey A. J. H., Kelly D. J. 2007. Fish community responses to drying disturbances in an intermittent stream: a landscape perspective. *Freshwater Biology* **52**: 1719-1733.
- Davey A. J. H., Kelly D. J., Biggs B. J. F. 2006. Refuge-use strategies of stream fishes in response to extreme low flows. *Journal of Fish Biology* **69**: 1047-1059.
- Death R. G. 1996. The effect of patch disturbance on stream invertebrate community structure: The influence of disturbance history. *Oecologia* **108**: 567-576.
- Death R. G., Collier K. J. 2010. Measuring stream macroinvertebrate responses to gradients of vegetation cover: When is enough enough? *Freshwater Biology* **55**: 1447-1464.
- Delong M. D., Brusven M. A. 1998. Macroinvertebrate community structure along the longitudinal gradient of an agriculturally impacted stream. *Environmental Management* **22**: 445-457.
- Didham R. K., Blakely T. J., Ewers R. M., Hitchings T. R., Ward J. B., Winterbourn M. J. 2012. Horizontal and vertical structuring in the dispersal of adult aquatic insects in a fragmented landscape. *Fundamental and Applied Limnology* **180**: 27-40.
- Dillaha T. A., Reneau R. B., Mostaghimi S., Lee D. 1989. Vegetative filter strips for agricultural nonpoint source pollution control. *Transactions of the American Society of Agricultural Engineers* **32**: 513-519.
- Dodds W. K., Oakes R. M. 2008. Headwater influences on downstream water quality. *Environmental Management* **41**: 367-377.
- Dole-Olivier M. J., Marmonier P., Befy J. L. 1997. Response of invertebrates to lotic disturbance: Is the hyporheic zone a patchy refugium? *Freshwater Biology* **37**: 257-276.
- Doppelt R. 1993. The vital role of the scientific community in the new river conservation strategies. *Journal of the North American Benthological Society* **12**: 189-193.
- Downes B. J., Keough M. J. 1998. Scaling of colonization processes in streams: Parallels and lessons from marine hard substrata. *Austral Ecology* **23**: 8-26.
- Downes B. J., Lake P. S., Glaister A., Bond N. R. 2006. Effects of sand sedimentation on the macroinvertebrate fauna of lowland streams: Are the effects consistent? *Freshwater Biology* **51**: 144-160.
- Elser P. 1999. Use of colonization baskets for the investigation of disturbance phenomena in streams under model conditions. *Limnologica* **29**: 120-127.
- Epps C. W., Wehausen J. D., Bleich V. C., Torres S. G., Brashares J. S. 2007. Optimizing dispersal and corridor models using landscape genetics. *Journal of Applied Ecology* **44**: 714-724.
- Erman D. C., Ligon F. K. 1988. Effects of discharge fluctuation and the addition of fine sediment on stream fish and macroinvertebrates below a water-filtration facility. *Environmental Management* **12**: 85-97.
- Gamboa M., Reyes R., Arrivillaga J. 2008. Benthic macroinvertebrates as bioindicators of environmental health. *Bulletin of Malariology and Environmental Health* **48**: 109-120.
- Goodwin C. N., Hawkins C. P., Kershner J. L. 1997. Riparian restoration in the Western United States: Overview and perspective. *Restoration Ecology* **5**: 4-14.

- Gray L. J., Ward J. V. 1982. Effects of sediment releases from a reservoir on stream macroinvertebrates. *Hydrobiologia* **96**: 177-184.
- Greenwood M. J., Harding J. S., Niyogi D. K., McIntosh A. R. 2012. Improving the effectiveness of riparian management for aquatic invertebrates in a degraded agricultural landscape: Stream size and land-use legacies. *Journal of Applied Ecology* **49**: 213-222.
- Gurnell A. M., Morrissey I. P., Boitsidis A. J., Bark T., Clifford N. J., Petts G. E., Thompson K. 2006. Initial adjustments within a new river channel: Interactions between fluvial processes, colonizing vegetation, and bank profile development. *Environmental Management* **38**: 580-596.
- Harding J., Clapcott J., Quinn J., Hayes J., Joy M., Storey R., Greig H., Hay J., James T., Beech M., Ozane R., Meredith A., Boothroyd I. 2009. *Stream habitat assessment protocols for wadeable rivers and streams of New Zealand*. Christchurch, New Zealand: School of Biological Sciences, University of Canterbury.
- Harding J. S. 2003. Historic deforestation and the fate of endemic invertebrate species in streams. *New Zealand Journal of Marine and Freshwater Research* **37**: 333-345.
- Harding J. S., Benfield E. F., Bolstad P. V., Helfman G. S., Jones E. B. D. 1998. Stream biodiversity: The ghost of land use past. *Proceedings of the National Academy of Sciences of the United States of America* **95**: 14843-14847.
- Harper D., Everard M. 1998. Why should the habitat-level approach underpin holistic river survey and management? *Aquatic Conservation-Marine and Freshwater Ecosystems* **8**: 395-413.
- Helfield J. M., Capon S. J., Nilsson C., Jansson R., Palm D. 2007. Restoration of rivers used for timber floating: Effects on riparian plant diversity. *Ecological Applications* **17**: 840-851.
- Herbst D. B., Bogan M. T., Roll S. K., Safford H. D. 2012. Effects of livestock exclusion on in-stream habitat and benthic invertebrate assemblages in montane streams. *Freshwater Biology* **57**: 204-217.
- Hildrew A. G., Woodward G., Winterbottom J. H., Orton S. 2004. Strong density dependence in a predatory insect: Large-scale experiments in a stream. *Journal of Animal Ecology* **73**: 448-458.
- Hobbs R. J. 2005. The future of restoration ecology: Challenges and opportunities. *Restoration Ecology* **13**: 239-241.
- Hobbs R. J., Norton D. A. 1996. Towards a conceptual framework for restoration ecology. *Restoration Ecology* **4**: 93-110.
- Hobbs R. J., Cramer V. A. 2008. Restoration ecology: Interventionist approaches for restoring and maintaining ecosystem function in the face of rapid environmental change. Report no. 15435938 (ISSN).
- Hoffmann A., Resh V. H. 2003. Oviposition in three species of limnephiloid caddisflies (Trichoptera): Hierarchical influences on site selection. *Freshwater Biology* **48**: 1064-1077.
- Hughes J. M. 2007. Constraints on recovery: Using molecular methods to study connectivity of aquatic biota in rivers and streams. *Freshwater Biology* **52**: 616-631.
- Hughes J. M., Schmidt D. J., Finn D. S. 2009. Genes in streams: Using DNA to understand the movement of freshwater fauna and their riverine habitat. *BioScience* **59**: 573-583.

- Huxel G. R., Hastings A. 1999. Habitat loss, fragmentation, and restoration. *Restoration Ecology* **7**: 309-315.
- Hylander K., Nilsson C., Jonsson B. G., Göthner T. 2005. Differences in habitat quality explain nestedness in a land snail meta-community. *Oikos* **108**: 351-361.
- Jähnig S. C., Lorenz A. W., Hering D. 2009. Restoration effort, habitat mosaics, and macroinvertebrates: Does channel form determine community composition? *Aquatic Conservation: Marine and Freshwater Ecosystems* **19**: 157-169.
- Jähnig S. C., Lorenz A. W., Hering D., Antons C., Sundermann A., Jedicke E., Haase P. 2011. River restoration success: A question of perception. *Ecological Applications* **21**: 2007-2015.
- Jansson R., Nilsson C., Malmqvist B. 2007. Restoring freshwater ecosystems in riverine landscapes: The roles of connectivity and recovery processes. *Freshwater Biology* **52**: 589-596.
- Jansson R., Nilsson C., Dynesius M., Andersson E. 2000. Effects of river regulation on river-margin vegetation: A comparison of eight boreal rivers. *Ecological Applications* **10**: 203-224.
- Kail J., Arle J., Jähnig S. C. 2012. Limiting factors and thresholds for macroinvertebrate assemblages in European rivers: Empirical evidence from three datasets on water quality, catchment urbanization, and river restoration. *Ecological Indicators* **18**: 63-72.
- Kail J., Hering D., Muhar S., Gerhard M., Preis S. 2007. The use of large wood in stream restoration: Experiences from 50 projects in Germany and Austria. *Journal of Applied Ecology* **44**: 1145-1155.
- Knapp R. A., Matthews K. R. 1996. Livestock grazing, golden trout, and streams in the golden trout wilderness, California: Impacts and management implications. *North American Journal of Fisheries Management* **16**: 805-820.
- Kreutzweiser D. P., Capell S. S., Good K. P. 2005. Macroinvertebrate community responses to selection logging in riparian and upland areas of headwater catchments in a northern hardwood forest. *Journal of the North American Benthological Society* **24**: 208-222.
- Lake P. S. 2000. Disturbance, patchiness, and diversity in streams. *Journal of the North American Benthological Society* **19**: 573-592.
- Lake P. S. 2003. Ecological effects of perturbation by drought in flowing waters. *Freshwater Biology* **48**: 1161-1172.
- Lake P. S., Bond N., Reich P. 2007. Linking ecological theory with stream restoration. *Freshwater Biology* **52**: 597-615.
- Lancaster J., Briers R. A., eds. 2008. *Aquatic insects: Challengers to populations* Wallingford: Royal Entomological Society of London.
- Lancaster J., Downes B. J., Arnold A. 2010. Oviposition site selectivity of some stream-dwelling caddisflies. *Hydrobiologia* **652**: 165-178.
- Larsen S., Vaughan I. P., Ormerod S. J. 2009. Scale-dependent effects of fine sediments on temperate headwater invertebrates. *Freshwater Biology* **54**: 203-219.
- Larsen S., Pace G., Ormerod S. J. 2011. Experimental effects of sediment deposition on the structure and function of macroinvertebrate assemblages in temperate streams. *River Research and Applications* **27**: 257-267.

- Larson M. G., Booth D. B., Morley S. A. 2001. Effectiveness of large woody debris in stream rehabilitation projects in urban basins. *Ecological Engineering* **18**: 211-226.
- Laycock W. A. 1991. Stable states and thresholds of range condition on North-American Rangelands: A viewpoint. *Journal of Range Management* **44**: 427-433.
- Lefrançois J., Grimaldi C., Gascuel-Oudou C., Gilliet N. 2007. Suspended sediment and discharge relationships to identify bank degradation as a main sediment source on small agricultural catchments. *Hydrological Processes* **21**: 2923-2933.
- Lenat D. R., Penrose D. L., Eagleson K. W. 1981. Variable effects of sediment addition on stream benthos. *Hydrobiologia* **79**: 187-194.
- Lepori F., Palm D., Brannas E., Malmqvist B. 2005. Does restoration of structural heterogeneity in streams enhance fish and macroinvertebrate diversity? *Ecological Applications* **15**: 2060-2071.
- Lindenmayer D. B., Fischer J., Felton A., Montague-Drake R., Manning A. D., Simberloff D., Youngentob K., Saunders D., Wilson D., Felton A. M., Blackmore C., Lowe A., Bond S., Munro N., Elliott C. P. 2007. The complementarity of single-species and ecosystem-oriented research in conservation research. *Oikos* **116**: 1220-1226.
- Lowrance R. R., Todd R. L., Asmussen L. E. 1984. Nutrient cycling in an agricultural watershed: I. Phreatic movement (Georgia, USA). *Journal of Environmental Quality* **13**: 22-27.
- Lyons J., Weigel B. M., Paine L. K., Undersander D. J. 2000. Influence of intensive rotational grazing on bank erosion, fish habitat quality, and fish communities in Southwestern Wisconsin trout streams. *Journal of Soil and Water Conservation* **55**: 271-276.
- Malmqvist B. 2000. How does wing length relate to distribution patterns of stoneflies (Plecoptera) and mayflies (Ephemeroptera)? *Biological Conservation* **93**: 271-276.
- Matthaei C. D., Piggott J. J., Townsend C. R. 2010. Multiple stressors in agricultural streams: Interactions among sediment addition, nutrient enrichment and water abstraction. *Journal of Applied Ecology* **47**: 639-649.
- Matthaei C. D., Weller F., Kelly D. W., Townsend C. R. 2006. Impacts of fine sediment addition to tussock, pasture, dairy and deer farming streams in New Zealand. *Freshwater Biology* **51**: 2154-2172.
- McGlone M. S. 1989. The Polynesian settlement of New Zealand in relation to environmental and biotic changes. *New Zealand Journal of Ecology* **12**: 115-129.
- McIntosh A. R. 2000. Habitat- and size-related variations in exotic trout impacts on native galaxiid fishes in New Zealand streams. *Canadian Journal of Fisheries and Aquatic Sciences* **57**: 2140-2151.
- McIntosh A. R., McHugh P. A., Dunn N. R., Goodman J. M., Howard S. W., Jellyman P. G., O'Brien L. K., Nystrom P., Woodford D. J. 2010. The impact of trout on galaxiid fishes in New Zealand. *New Zealand Journal of Ecology* **34**: 195-206.
- Miller J. R., Hobbs R. J. 2007. Habitat restoration: Do we know what we're doing? *Restoration Ecology* **15**: 382-390.

- Miller S. W., Budy P., Schmidt J. C. 2010. Quantifying macroinvertebrate responses to in-stream habitat restoration: Applications of meta-analysis to river restoration. *Restoration Ecology* **18**: 8-19.
- Milner A. M., Knudsen E. E., Soiseth C., Robertson A. L., Schell D., Phillips I. T., Magnusson K. 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.
- Muotka T., Syrjänen J. 2007. Changes in habitat structure, benthic invertebrate diversity, trout populations and ecosystem processes in restored forest streams: A boreal perspective. *Freshwater Biology* **52**: 724-737.
- Neboiss A. 1986. Atlas of Trichoptera of the SW Pacific - Australian region. Boston: W Junk Publishers.
- Nilsson C., Lepori F., Malmqvist B., Törnlund E., Hjerdt N., Helfield J. M., Palm D., Östergren J., Jansson R., Brännäs E., Lundqvist H. 2005. Forecasting environmental responses to restoration of rivers used as log floatways: An interdisciplinary challenge. *Ecosystems* **8**: 779-800.
- Owens L. B., Edwards W. M., Van Keuren R. W. 1996. Sediment losses from a pastured watershed before and after stream fencing. *Journal of Soil and Water Conservation* **51**: 90-94.
- Owens P. N., Batalla R. J., Collins A. J., Gomez B., Hicks D. M., Horowitz A. J., Kondolf G. M., Marden M., Page M. J., Peacock D. H., Peticrew E. L., Salomons W., Trustrum N. A. 2005. Fine-grained sediment in river systems: Environmental significance and management issues. *River Research and Applications* **21**: 693-717.
- Palmer M. A., Filoso S. 2009. Restoration of ecosystem services for environmental markets. *Science* **325**: 575-576.
- Palmer M. A., Bely A. E., Berg K. E. 1992. Response of invertebrates to lotic disturbance: A test of the hyporheic refuge hypothesis. *Oecologia* **89**: 182-194.
- Palmer M. A., Allan J. D., Butman C. A. 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., Ambrose R. F., Poff N. L. 1997. Ecological theory and community restoration ecology. *Restoration Ecology* **5**: 291-300.
- Palmer M. A., Menninger H. L., Bernhardt E. 2010. River restoration, habitat heterogeneity and biodiversity: A failure of theory or practice? *Freshwater Biology* **55**: 205-222.
- Palmer M. A., Bernhardt E. S., Allan J. D., Lake P. S., Alexander G., Brooks S., Carr J., Clayton S., Dahm C. N., Follstad Shah J., Galat D. L., Loss S. G., Goodwin P., Hart D. D., Hassett B., Jenkinson R., Kondolf G. M., Lave R., Meyer J. L., O'Donnell T. K., Pagano L., Sudduth E. 2005. Standards for ecologically successful river restoration. *Journal of Applied Ecology* **42**: 208-217.

- Parkyn S. M., Davies-Colley R. J., Halliday N. J., Costley K. J., Croker G. F. 2003. Planted riparian buffer zones in New Zealand: Do they live up to expectations? *Restoration Ecology* **11**: 436-447.
- Peckarsky B. L., Taylor B. W., Caudill C. C. 2000. Hydrologic and behavioral constraints on oviposition of stream insects: Implications for adult dispersal. *Oecologia* **125**: 186-200.
- Petersen I., Masters Z., Hildrew A. G., Ormerod S. J. 2004. Dispersal of adult aquatic insects in catchments of differing land use. *Journal of Applied Ecology* **41**: 934-950.
- Petticrew E. L., Krein A., Walling D. E. 2007. Evaluating fine sediment mobilization and storage in a gravel-bed river using controlled reservoir releases. *Hydrological Processes* **21**: 198-210.
- Purcell A. H., Friedrich C., Resh V. H. 2002. An assessment of a small urban stream restoration project in Northern California. *Restoration Ecology* **10**: 685-694.
- Quinn J. M., Hickey C. W. 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., Boothroyd I. K. G., Smith B. J. 2004. Riparian buffers mitigate effects of pine plantation logging on New Zealand streams: 2. Invertebrate communities. *Forest Ecology and Management* **191**: 129-146.
- Quinn J. M., Cooper A. B., Davies-Colley R. J., Rutherford J. C., Williamson R. B. 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.
- Rabeni C. F., Doisy K. E., Zweig L. D. 2005. Stream invertebrate community functional responses to deposited sediment. *Aquatic Sciences* **67**: 395-402.
- Rabeni C. F., Doisy K. E., Zweig L. D. 2005. Stream invertebrate community functional responses to deposited sediment. *Aquatic Sciences* **67**: 395-402.
- Rae J. G. 1987. The effects of flooding and sediments on the structure of a stream midge assemblage. *Hydrobiologia* **144**: 3-10.
- Reice S. R., Wissmar R. C., Naiman R. J. 1990. Disturbance regimes, resilience, and recovery of animal communities and habitats in lotic ecosystems. *Environmental Management* **14**: 647-659.
- Reich P., Downes B. J. 2003. Experimental evidence for physical cues involved in oviposition site selection of lotic hydrobiosid caddis flies. *Oecologia* **136**: 465-475.
- Reich P., Downes B. J. 2004. Relating larval distributions to patterns of oviposition: Evidence from lotic hydrobiosid caddisflies. *Freshwater Biology* **49**: 1423-1436.
- Renöfält B. M., Jansson R., Nilsson C. 2005. Spatial patterns of plant invasiveness in a riparian corridor. *Landscape Ecology* **20**: 165-176.
- Rhodes H. M., Closs G. P., Townsend C. R. 2007. Stream ecosystem health outcomes of providing information to farmers and adoption of best management practices. *Journal of Applied Ecology* **44**: 1106-1115.
- Rohde S., Schütz M., Kienast F., Englmaier P. 2005. River widening: An approach to restoring riparian habitats and plant species. *River Research and Applications* **21**: 1075-1094.

- Roni P., Quinn T. P. 2001. Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams. *Canadian Journal of Fisheries and Aquatic Sciences* **58**: 282-292.
- Roni P., Hanson K., Beechie T. 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. *North American Journal of Fisheries Management* **28**: 856-890.
- Roni P., Beechie T. J., Bilby R. E., Leonetti F. E., Pollock M. M., Pess G. R. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific northwest watersheds. *North American Journal of Fisheries Management* **22**: 1-20.
- Roni P., Bennett T., Morley S., Pess G. R., Hanson K., Van Slyke D., Olmstead P. 2006. Rehabilitation of bedrock stream channels: The effects of boulder weir placement on aquatic habitat and biota. *River Research and Applications* **22**: 967-980.
- Siva-Jothy M. T., Wingfield Gibbons D., Pain D. 1995. Female oviposition-site preference and egg hatching success in the damselfly *Calopteryx splendens xanthostoma*. *Behavioral Ecology and Sociobiology* **37**: 39-44.
- Smith B. J., Collier K. J., Halliday N. J. 2002. Composition and flight periodicity of adult caddisflies in New Zealand hill-country catchments of contrasting land use. *New Zealand Journal of Marine and Freshwater Research* **36**: 863-878.
- Smith R. F., Alexander L. C., Lamp W. O. 2009. Dispersal by terrestrial stages of stream insects in urban watersheds: a synthesis of current knowledge. *Journal of the North American Benthological Society* **28**: 1022-1037.
- Sode A., Wieberg-Larsen P. 1993. Dispersal of adult Trichoptera at a Danish forest brook. *Freshwater Biology* **30**: 439-446.
- Spänhoff B., Arle J. 2007. Setting attainable goals of stream habitat restoration from a macroinvertebrate view. *Restoration Ecology* **15**: 317.
- Spänhoff B., Riss W., Jäkel P., Dakkak N., Meyer E. I. 2006. Effects of an experimental enrichment of instream habitat heterogeneity on the stream bed morphology and chironomid community of a straightened section in a sandy lowland stream. *Environmental Management* **37**: 247-257.
- 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.
- Stone M. L., Whiles M. R., Webber J. A., Williard K. W. J., Reeve J. D. 2005. Macroinvertebrate communities in agriculturally impacted Southern Illinois streams: Patterns with riparian vegetation, water quality, and in-stream habitat quality. *Journal of Environmental Quality* **34**: 907-917.
- Storey R. G., Quinn J. M. 2011. Life histories and life history strategies of invertebrates inhabiting intermittent streams in Hawke's Bay, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **45**: 213-230.
- Strum C., Barth G. 2012. Sustaining Colorado Watershed Conference 2011.

- Sudduth E. B., Hassett B. A., Cada P., Bernhardt E. S. 2011. Testing the field of dreams hypothesis: Functional responses to urbanization and restoration in stream ecosystems. *Ecological Applications* **21**: 1972-1988.
- Suding K. N., Hobbs R. J. 2009. Threshold models in restoration and conservation: A developing framework. *Trends in Ecology and Evolution* **24**: 271-279.
- Suding K. N., Gross K. L., Houseman G. R. 2004. Alternative states and positive feedbacks in restoration ecology. *Trends in Ecology and Evolution* **19**: 46-53.
- Sundermann A., Stoll S., Haase P. 2011. River restoration success depends on the species pool of the immediate surroundings. *Ecological Applications* **21**: 1962-1971.
- Suren A. M., McMurtrie S. 2005. Assessing the effectiveness of enhancement activities in urban streams: II. Responses of invertebrate communities. *River Research and Applications* **21**: 439-453.
- Tank J. L., Winterbourn M. J. 1996. Microbial activity and invertebrate colonisation of wood in a New Zealand forest stream. *New Zealand Journal of Marine and Freshwater Research* **30**: 271-280.
- Taylor S. L., Roberts S. C., Walsh C. J., Hatt B. E. 2004. Catchment urbanisation and increased benthic algal biomass in streams: Linking mechanisms to management. *Freshwater Biology* **49**: 835-851.
- Törnblom J., Roberge J. M., Angelstam P. 2011. Rapid assessment of headwater stream macroinvertebrate diversity: An evaluation of surrogates across a land-use gradient. *Fundamental and Applied Limnology* **178**: 287-300.
- Townsend C. R., Downes B. J., Peacock K., Arbuckle C. J. 2004. Scale and the detection of land-use effects on morphology, vegetation and macroinvertebrate communities of grassland streams. *Freshwater Biology* **49**: 448-462.
- Townsend C. R., Doledec S., Norris R., Peacock K., Arbuckle C. 2003. The influence of scale and geography on relationships between stream community composition and landscape variables: Description and prediction. *Freshwater Biology* **48**: 768-785.
- Trimble S. W., Mendel A. C. 1995. The cow as a geomorphic agent: A critical review. *Geomorphology* **13**: 233-253.
- Urban M. C., Skelly D. K., Burchsted D., Price W., Lowry S. 2006. Stream communities across a rural-urban landscape gradient. *Diversity and Distributions* **12**: 337-350.
- Verberk W. C. E. P., Sipel H., Esselink H. 2008. Life-history strategies in freshwater macroinvertebrates. *Freshwater Biology* **53**: 1722-1738.
- Violin C. R., Cada P., Sudduth E. B., Hassett B. A., Penrose D. L., Bernhardt E. S. 2011. Effects of urbanization and urban stream restoration on the physical and biological structure of stream ecosystems. *Ecological Applications* **21**: 1932-1949.
- Vörösmarty C. J., McIntyre P. B., Gessner M. O., Dudgeon D., Prusevich A., Green P., Glidden S., Bunn S. E., Sullivan C. A., Liermann C. R., Davies P. M. 2010. Global threats to human water security and river biodiversity. *Nature* **467**: 555-561.
- Ward J. V., Tockner K. 2001. Biodiversity: Towards a unifying theme for river ecology. *Freshwater Biology* **46**: 807-819.

- Williams D. D., Hynes H. B. N. 1976. The recolonization mechanisms of stream benthos. *Oikos* **27**: 265-272.
- Winterbourn M. J. 2007. Stable isotope analysis identifies morphologically indistinguishable caddisflies. *New Zealand Journal of Marine and Freshwater Research* **41**: 401-404.
- Winterbourn M. J., Crowe A. L. M. 2001. Flight activity of insects along a mountain stream: Is directional flight adaptive? *Freshwater Biology* **46**: 1479-1489.
- Winterbourn M. J., Gregson K. L. D., Dolphin C. H. 2006. *Guide to the Aquatic Insects of New Zealand*: The Entomological Society of New Zealand
- Winterbourn M. J., Chadderton W. L., Entekin S. A., Tank J. L., Harding J. S. 2007. Distribution and dispersal of adult stream insects in a heterogeneous montane environment. *Fundamental and Applied Limnology* **168**: 127-135.
- Wohl E., Angermeier P. L., Bledsoe B., Kondolf G. M., MacDonnell L., Merritt D. M., Palmer M. A., Poff N. L., Tarboton D. 2005. River restoration. *Water Resources Research* **41**: 157-169.
- Wood P. J., Armitage P. D. 1997. Biological effects of fine sediment in the lotic environment. *Environmental Management* **21**: 203-217.
- Young R. G., matthaei C. D., Townsend C. R. 2008. Organic matter breakdown and ecosystem metabolism: Functional indicators for assessing river ecosystem health. . *Journal of the North American Benthological Society* **27**: 605-625.