STREAM REHABILITATION WITHIN AGRICULTURAL RIVERSCAPES

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Abstract

Modern agriculture is necessary to feed our growing global population, yet agriculture is also the most extensive cause of freshwater ecosystem degradation. Accordingly, determining how to maintain or improve freshwater values within intensively farmed riverscapes is a key environmental challenge facing society. Broadly, my PhD thesis aims to consolidate and create knowledge to make the outcomes of stream rehabilitation in agricultural landscapes more predictable. More specifically, my aims are to 1) develop practical habitat data collection and rehabilitation planning methods to improve stream fisheries, and 2) determine outcomes for fish species resulting from rehabilitation projects set within agricultural catchments.

To this end, in Chapter 2, I review the salmonid literature to identify key attributes that can limit stream brown trout populations. I also determine numerical thresholds for stream attributes that are likely to support various trout population levels—from non-existent to thriving. This literature review (presented in Appendix 1) demonstrates there are a wide range of attributes that should be considered in order to achieve community aspirations for productive stream fisheries. The literature review underpins a Bayesian Belief Network-based decision support model (BBN). This BBN directs users to assemble a parsimonious environmental data set to inform stream fishery management. It also integrates and interrogates these data to generate standardised and testable hypotheses about which environmental factors are likely to limit trout productivity. I tested the BBN on the Horokiri Stream, a data-rich catchment in Wellington, New Zealand. The BBN results suggest that the fishery was recruitment-limited in its reference state and limited by cover and low summer-flows in its degraded state. These model results were comparable with the conclusions of five experienced fishery biologists, following their detailed investigation into the factors that led to the loss of the Horokiri trout fishery between 1951 and 1990. Chapter 2 demonstrates the suitability of BBN modeling for conducting a limiting-factor analysis on stream fish.

Mechanically reshaping stream banks is a common practice to mitigate bank erosion in streams that have been channelised and lowered for agricultural land drainage. However, the response of fish populations to this practice has rarely been quantitatively evaluated. In Chapter 3, I assess the fish and habitat responses to a catchment-scale bank reshaping event in Waituna Creek (Southland, New Zealand), a low-gradient stream that drains an intensive agricultural landscape. Fish and instream habitat data
were collected before and annually for three years after the reshaping event using a Before-After-Control-Impact study design. I hypothesised that large-bodied fish, such as eels and trout, would be negatively affected by the practice. After reshaping, deposited fine sediment levels increased in impact reaches and there was also a significant reduction in longfin eel biomass (by 49%). Three years after reshaping, fish community structure had largely returned to its pre-impact state in the reshaped areas. These results show that stream bank reshaping can have substantial effects on fish populations, even in streams which are subject to regular mechanical disturbance.

In Chapter 4, I use a space-for-time substitution design to investigate the response of instream habitat and fish populations to different riparian management practices throughout the Waikakahi Stream—a Dairy Best Practice Catchment in South Canterbury (New Zealand). I found a significant negative correlation between the upstream area of stock exclusion fencing and deposited instream fine sediment cover. Furthermore, I determined that this relationship emerges when ≥ 300 m lengths of upstream riparian area were included in the analysis—indicating the scale at which stock exclusion fencing results in a desirable instream habitat response. This result shows that if a catchment-scale approach is taken to improving riparian habitat, with broad landowner buy-in and determined community leadership, then some instream habitat improvement can be achieved within just a decade.

In my General Discussion (Chapter 5), I outline the wider implications of my research with specific reference to contemporary freshwater management in New Zealand. I also present a stream management ‘thought experiment’ and a new conceptual model that provide theoretical frameworks for considering how spatiotemporal scale might influence the outcomes of stream rehabilitation. Overall, my PhD research highlights the importance of considering a broad range of attributes when assessing the ecological effects of habitat degradation or rehabilitation on instream habitat and fish. In particular, physical instream habitat attributes such as structural fish cover can limit fish populations in agricultural streams, even if they have relatively degraded water quality. Successful stream fish population management in New Zealand will require incorporating physical/structural habitat attributes into the relevant management frameworks.
Dedication

For the fish in the streams and for my parents who fostered my curiosity and drive to discover their secrets.
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List of thesis publications

This thesis was written by publication. Chapters 2, 3 and 4 have been published in peer reviewed scientific journals. In addition, Appendix 1 has been published as a Cawthron Institute technical report and. The references for these publications are as follows:

Chapter 2:


Chapter 2 (Appendix 1):


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1. General introduction

The phrase ‘intensive agriculture’ has become synonymous with degraded freshwater ecosystems—for good reason. Globally, there is overwhelming evidence implicating agriculture as the most extensive cause of freshwater ecosystem stress (Allan 2004; Vorosmarty 2010). Increased fine sediment and nutrient inputs from agricultural runoff are particularly pervasive, non-point source pollutants (Parkyn et al. 2003; Allan 2004). These stressors occur alongside other physicochemical alterations to surface and ground water, flow abstraction as well as modifications to the river habitat template to enable land drainage and manage floods (Carpenter et al. 1998; Mosley 1998, Chambers et al. 2006, Blann et al. 2009). When combined, all these stressors alter ecosystem structure and function, reduce biodiversity and diminish ecosystem services (Townsend et al. 2008). The complexity of interacting stressor-response relationships across different spatial, temporal and administrative scales has created a truly wicked problem for managing agricultural catchments (DeFries & Nagendra 2017).

Yet modern agriculture is necessary, for the obvious reason that it is vital for economies such as New Zealand. And of course, a growing global human population needs feeding (MFE 2001; Foote et al. 2015). A key challenge facing society is finding ways to maintain, improve or restore freshwater ecosystems, and the values they support, within intensively farmed riverscapes (Tilman et al. 2002). In response to this challenge, farm environmental practices have undergone substantive industry-led improvements over the past three decades (Dale & Polasky 2007; Prokopy et al. 2008; Melland et al. 2018). Despite these changes, many freshwater ecosystems remain in a degraded state as a result of current practices, continuing agricultural development, or legacy effects of past land use (Harding et al. 1998; Meals et al. 2009). Consequently, the desire for more effective intervention is growing, particularly as tangible freshwater values, such as fisheries, are diminished to levels deemed unacceptable by communities (Stewart et al. 2009). To meet this need, aquatic restoration and rehabilitation has become a growing domain of scientific inquiry.
1.1. River restoration and ecological theory

Although still relatively young, the science of river restoration ecology is firmly grounded within an established body of theory describing how ecosystems respond to stress. Highly relevant concepts include disturbance theory (e.g. Resh et al. 1988; Townsend et al. 1998), multiple stress ecology (Ormerod et al. 2010), ecosystem resistance and resilience (Allison 2004; Lake 2013, Hilderbrand & Utz 2015), the river scales concept (Frissel et al. 1986, Townsend 1996), and population dynamics (e.g. Turchin 2001). However, it is worth noting that the entire canon of general ecological theory will be relevant to aquatic restoration in certain circumstances.

The simplistic assumption underlying restoration attempts is the notion that if anthropogenic disturbance is removed (through a restoration action), ecosystems will re-establish some or all of their original structure and functions. However, the state and functions that characterise a degraded ecosystem are the result of disturbance regimes and multiple stressors acting and interacting, sometimes in complex non-additive ways, across broad space and time scales (Townsend et al. 2008; Meals et al. 2009; Matthaei et al. 2010, Vorosmarty 2010). Moreover, the fate of ecosystems and populations are in part dependent on antecedent conditions and these legacy effects interact with any contemporary mix of pressures (Harding et al. 1998). All this complexity means we should expect ecosystems to frequently respond to restoration attempts in surprising or (apparently) chaotic ways (Magnuson 1990; Lake 2007; Scheffer et al. 2001). Nevertheless, progress within theoretical ecology has improved our predictive capabilities. Below I briefly explain some key ecological tenets that should be considered when designing restoration initiatives for rivers.

Bender et al. (1984) and Lake et al. (2000), usefully categorise ecosystem disturbance as either:
1) Long term ‘press’ disturbances, for example, the diffuse leaching of excessive nutrients from farmland into waterbodies,

2) Short term ‘pulse’ disturbances, for example, a large-scale flood event or the periodic discharge of pollutants, or

3) Escalating ‘ramp’ disturbances, for example, the increasing stress that results from water abstraction during persistent drought.

In rivers, all these disturbance types may operate with overlapping periodicity and distributions. Alongside natural disturbance, anthropogenic disturbance regimes act as environmental filters to produce the observed set of species, functions and processes within a river ecosystem (e.g. Woodward & Diament 1991). The long-term success of any river restoration effort requires first identifying the types of stress that comprise the overriding anthropogenic disturbance regime and then reducing the severity, extent and/or duration of the most damaging aspects of that regime. Alternatively, or in addition, restoration efforts can focus on increasing an ecosystem’s resistance and/or resilience to disturbance. For example, connecting fragmented habitats has long been a central tenant within terrestrial restoration ecology and is being applied increasingly within river restoration contexts (Seliger & Zeiringer 2018). As populations become increasingly connected across a wider geographic area they are more able to recolonise perturbed areas from connected habitats, thus becoming more resilient.

The concepts of ecosystem resistance and resilience describe how relatively stable ecosystem structures and functions persist under disturbance regimes (Allison 2004; Lake 2013, Hildebrand & Utz 2015). Ecosystem resistance is the ability of system components to withstand disturbance or stress, whereas, ecosystem resilience is the ability of an ecosystem to recover from stress (e.g. through recolonisation or population growth). ‘Negative’ resistance and resilience is the same concept but in reverse. That is, inertia must be overcome before an ecosystem regains lost structural or functional components in response to the removal of stress (Lake 2013). In some cases, an ecosystem may display extreme
resistance to recovery, for example, because of the loss of a keystone species. Alternatively, an ecosystem may become highly destabilised during recovery, or, recovery may be subject to substantial lag periods see: Sarr (2002) and Lake’s (2007) depiction of the ‘humpty dumpty’, ‘shifting target’ and ‘hysteresis’ recovery models, respectively.

Consideration of the scale of disturbance regimes is vital for predicting how ecosystems respond to restoration efforts. Frissel et al. (1986) described rivers as a nested hierarchy of spatial scales, where stream patches (<1m²) are nested within stream reaches (10-1000m), which in turn are nested within river segments (1000 – 10,000m) and catchments (e.g. >10³km²). In general, features and processes operating at larger scales affect smaller scales but not vice versa. For example, a catchment’s underlying geology will profoundly affect reach-scale channel morphology but geology is primarily driven by larger-scale, longer-term processes (such as plate tectonics). Hynes (1975) put this concept more elegantly, when he posited that “in every respect, the valley rules the stream”. If successful restoration depends on addressing the overriding anthropogenic disturbance regime(s), and most rivers are degraded by catchment-scale processes such as land use intensification (Allan 2004), then the nested nature of river ecosystems implies that patch to segment scale restoration attempts may be destined to fail in extensively developed catchments.

Some river fauna can use entire river catchments and beyond for different phases of growth and reproduction (e.g. diadromous salmonids). Therefore, knowledge of population dynamics and life histories can be critical to planning successful restoration (Alldredge et al. 2015). When attempting ecosystem restoration for individual species, or groups of species, there are two key concepts that need careful consideration: 1) the population carrying capacity of the environment, and 2) the related concepts of density-dependent and density-independent growth and survival. Carrying capacity is the term used to describe the upper limit to the
number of organisms that an ecosystem can sustain. The upper limit of any river population is not static but changes over time in response to limiting factors such as food and space availability, particularly in rivers with highly variable flow regimes (Cramer & Ackerman 2009). As any population approaches carrying capacity, density-dependent factors increasingly constrain population growth until carrying capacity is reached. The logistic function in Figure 1 illustrates this concept.

Figure 1. A typical logistic curve used in stock recruitment models. Initially, a population at low densities undergoes exponential growth. Eventually, population growth is increasingly constrained as carrying capacity is approached.

Food limitation is an intuitive and common density-dependent constraint on population growth. However, breeding habitat availability, juvenile rearing habitat, predation pressure and disease are other types of density-dependent controls that can increase in severity at high population densities (Armstrong et al. 2003). Inverse density-dependent controls can also occur at very low population sizes. For example, the chances of finding a mate may be reduced at very low population densities and this can further reduce population size (Courchamp et al. 1999). Typical examples of density-independent population controls are the effect of high temperatures or flooding in rivers (George et al.
These factors are thought to slow growth or ‘thin-out’ a proportion of river fauna irrespective of initial population size. Restoration actions that focus on increasing the amount of food or space, for example by increasing the minimum flow of a river, can be expected to be relatively ineffective if a population is maintained well below carrying capacity through density independent constraints (such as regular and intense flooding).

In my thesis I combine the concepts of density-dependent and density-independent limiting factors within a single model that predicts how interacting stressors and environmental factors can limit stream fish populations (Chapter 2). I also show empirically how physical habitat and fish populations respond to catchment scale disturbance and restoration efforts in farm streams (Chapter 3 and 4). My results support the notion that for stream restoration to be effective, multiple ecosystem pressures need to be considered simultaneously. In addition, the scale of restoration effort should match the scale at which degradation occurs. I build on this concept in my final chapter (Chapter 5) by exploring how the idea of nested river-scales can be incorporated into predictions of river ecosystem recovery.

Above I have briefly described a few of the ecological theories that can be brought to bare within river restoration science. However, in practice, paying due diligence to ecological theory is seldom achievable because of political, economic or logistic constraints (Lake et al. 2007). Below I discuss the realities of applied river restoration both abroad and within New Zealand.

1.2. River restoration in practice

Freshwater habitat restoration is now a multibillion-dollar industry globally (Bernhardt et al. 2005). Vast resources and science investment are devoted to habitat improvements to benefit ailing salmonid stocks in the Pacific Northwest (Roni et al. 2008; Stewart et al. 2009). Large restoration projects are also
commonplace across Europe and Asia (Roni et al. 2008). Broadly, agricultural river restoration actions comprise ‘passive measures’, such as livestock exclusion from riparian areas and pastural land retirement/reforestation, or, ‘active measures’ such as reinstating channel meanders, bank contouring, instream structural habitat addition, fish passage improvement, and deposited fine sediment removal (Kauffman et al. 1997; Pretty et al. 2003; Lake 2007).

Generally, active rehabilitation measures are undertaken in an attempt to fast-track the development of more complex instream habitat structures and processes. The assumption being that creating more complex reach- and segment-scale structural and hydraulic habitats will enable a waterbody to support a more diverse range of species and ecosystem functions (Rohr et al. 2018). However, numerous literature reviews of freshwater restoration projects show that the responses of target biota to active restoration measures are often underwhelming (Kondolf & Micheli 1995; Bash & Ryan 2002; Parkyn et al. 2003; Bernhardt et al. 2005, Hillman & Brierley 2005; Lester & Boulton 2008; Stewart et al. 2009). This is because restoration projects frequently fail to address overriding catchment-scale processes, such as non-point source water pollution or degradation of remote habitats that are crucial for a target organism’s life history. For example, a salmonid population may be unable to repopulate engineered rearing habitat because headwater spawning areas may remain in a degraded state. Perhaps a more mundane reason for the apparent failure of many restoration projects is that restoration monitoring is seldom applied across the appropriate space and time scales to detect a biotic response—potentially resulting in systematic false-negative reporting (Palmer et al. 2005). Nevertheless, some projects are at least partially successful at improving ecosystem health (Palmer et al. 2005; Summers 2010; Palmer et al. 2014; Keesstra et al. 2018).

International authors have repeatedly highlighted a need for more monitoring to determine the success (and practicality) of various stream restoration actions (Kondolf & Micheli 1995; Kauffman et al. 1997; Roni et al. 2002; Hillman & Brierley 2005; Beechie et al. 2008; Michel et al. 2014). River and stream restoration
resources tend to be exhaustively invested into on-the-ground action, with little or no resources left for monitoring (Watzin & McIntosh 1999; Beechie et al. 2008; Roni et al. 2008; Leathwick et al. 2009; Bernhardt & Palmer 2011). Project ‘success’ is often determined by an optimistic interpretation of ecological theory with respect to an expected biotic response to physical habitat changes brought about by restoration efforts.

1.3. River restoration in New Zealand

New Zealand has recently started catchment-scale river (and lake) restoration, following the path taken by the more populous developed nations. However, unlike the continents, investment in freshwater restoration is still comparatively modest. So far, excluding livestock from waterbodies and re-establishing native riparian vegetation (passive measures) are the only restoration initiatives that have occurred at a landscape scale in farmed catchments (Parkyn et al. 2003; Dodd et al. 2009; Greenwood et al. 2012; Wahl et al. 2013; Mckergow et al. 2016). Although these actions are wide-spread across New Zealand (Peters et al. 2015), there has been little attempt to investigate any ecological outcomes. Exceptions include some well-resourced studies that have determined water quality responses to good farming practices in selected small- to moderate-sized streams under the Dairy Best Practice Catchment programme (Monaghan et al. 2009). Significant reductions in *Escherichia coli* (*E. coli*), phosphorus and suspended fine sediment loads have been documented (Wilcock et al. 1999; Wilcock et al. 2007; Collier et al. 2009; Davies-Colley et al. 2010; Collins et al. 2012). A few studies have also investigated benthic macroinvertebrate community responses, but these generally report equivocal results (Storey and Cowley 1997; Quinn 2009; Parkyn et al. 2003; Graham et al. 2018).

Less is known about the effects of farm stream restoration on freshwater fishery species in New Zealand. Brown trout (*Salmo trutta*) and anguillid eels support highly valued global fisheries. In New Zealand, longfin and shortfin eels (*Anguilla*
dieffenbachii and Anguilla australis) are a taonga species—integral to Maori culture as mahinga kai throughout both North and South Islands (McDowall 2011). New Zealand’s brown trout populations support internationally renowned recreational fisheries, particularly in the South Island (McDowall 1994). Improving the ability of waterbodies to provide for recreational fisheries or mahinga kai is a common motivation for restoration projects in New Zealand and elsewhere (Monaghan et al. 2009; Collier 2017). For example, the provision of plentiful mahinga kai that is safe to eat is a key objective of the Waikato River Authority restoration programme (WRA 2011). Yet we do not know how trout and eel populations respond to our current approaches to restoration.

Even less is known of the value of active stream rehabilitation actions in New Zealand, where there has been almost no scientific attention. David & Hamer (2012) and David et al. (2014) show that targeted native fish passage remediation can be very successful. However, other potential stream habitat manipulation practices that could have positive or negative effects on fish populations have not been formally investigated. For example, no research has been published on the effects of channel reconfiguration, to reduce bank erosion, on New Zealand fish populations (Hudson & Harding 2004), although there are ongoing field trials by the University of Canterbury’s Carex Team (see https://www.canterbury.ac.nz/science/schools-and-departments/biological-sciences/research/ferg/carex/). Clearly, there are substantial knowledge gaps on the effects of both active and passive stream restoration actions on freshwater fish populations in New Zealand.

1.4. Thesis aims and structure

Broadly, my PhD thesis aims to consolidate existing, and create new knowledge, to enable more effective rehabilitation of native and introduced salmonid fishery values in intensified agricultural streams. More specifically, the aims of the thesis are to 1) develop cost-efficient rehabilitation planning and monitoring methods, and 2) determine fish population outcomes resulting from stream rehabilitation
projects set within farmed riverscapes. Below I briefly outline the individual thesis chapters with a focus on the chapter-specific aims and hypotheses.

In Chapter 2, I provide a substantial review of the salmonid literature (which is presented as a separate body of work in Appendix 1) and consolidate this information within a Bayesian Belief Network (BBN) model. The BBN model enables a systematic approach to determine potential limiting factors for a stream brown trout population. A range of pressures (e.g. pollutants, fishing pressure, and habitat quantity / quality) are considered and the critical environmental factor(s) likely to be limiting the productivity of a stream fishery are identified—once the data requirements are met and entered into the BBN. I test the BBN model performance using a historical data set from the (once) notable trout fishery in the Horokiri Stream (Wellington).

In Chapter 3, I assess the fish community response to a catchment-scale mechanical bank reshaping event in Waituna Creek (Southland) using a Before-After-Control-impact (repeated measures) study design. The bank reshaping was undertaken by the regional authority in an effort to reduce fine sediment entering the highly valued Waituna Lagoon downstream. Because bank-edge fish cover is removed during reshaping, I hypothesise that populations of large-bodied fish with a strong affinity for cover (i.e. eels and trout) will be negatively affected by the reshaping event.

Chapter 4 describes a catchment-scale survey of riparian condition, in-stream habitat and trout and eel population data in the Waikakahi Stream (South Canterbury). I interrogate these data for correlations between on-farm riparian management practices and in-stream habitat and fish responses. I predict that trout and eel densities and biomass will be higher in stream reaches adjacent to well managed riparian areas (i.e. wide stock exclusion areas with mature riparian
vegetation). In addition, I hypothesise that riparian habitat quality indicators will correlate with instream habitat quality indicators at the reach to segment scales.

In the General Discussion (Chapter 5), I consolidate the implications of the individual thesis chapters and explain how stream restoration can fit within contemporary New Zealand freshwater management frameworks. In addition, I discuss how the concept of ecosystem hysteresis (Scheffer et al. 2001; Beisner et al. 2003; Ratajczak et al. 2018) might apply within a river restoration context. This is done with a ‘thought experiment’ and a standalone review piece written in the form of a short scientific communication. I also describe a new conceptual model that predicts how spatial scale might influence river ecosystem recovery following the removal of stressors.

1.5. A note on semantics

1.5.1. Restoration or rehabilitation?
The terms ‘ecosystem restoration’ and ‘ecosystem rehabilitation’ are used, often interchangeably, throughout the literature. However, each has a fundamentally different goal. Restoration attempts to reinstate aspects of some perceived preexisting ecosystem condition, often analogous to a historical pre-industrial state. Restoration can be considered an attempt to reinstate a more natural ecosystem. In contrast, ecosystem rehabilitation (sometimes termed ‘reconstruction’) does not require that ecosystems are set on a trajectory towards their natural state; rather, their existing state is improved according to human values or for the conservation of desirable or rare species and/or ecosystems (Callicott 2011; Sarkar 2011). For example, rehabilitation could include modifying a degraded environment to support a fishery for an introduced species (e.g. brown trout in New Zealand). In this instance, rehabilitation has a goal in opposition of restoration, because the presence of an introduced species is a departure from an ecosystem’s natural state and introduced trout in particular can negatively impact native fauna (McDowall 2006). Within the rest of this thesis, I focus on rehabilitation. This more pragmatic approach to ecosystem improvement is less
restrictive, and therefore, arguably more suitable for highly degraded farm streams, where socioeconomic constraints may prevent a return to a state even remotely resembling natural conditions. The rehabilitation approach also avoids the logical inconsistency of attempting to restore an ecosystem to a perceived reference ‘state’, when in fact, all ecosystems are dynamic and constantly changing entities over space and time (Sarkar 2011).

1.5.2. The use of the personal pronoun in this thesis

The following three chapters (Chapters 2, 4 and 4) are published in peer reviewed journals and Appendix 1 has been published as a Cawthron Institute technical report. All the work described in this thesis is based on work done by me as the lead author. However, I received editorial and theoretical input from my supervisors and colleagues including: John Hayes, Christoph Matthaei, Gerry Closs, Rasmus Gabrielsson and others. Accordingly, I have referred to work being done individually as "we" instead of "I" to remain consistent with the publications that are already within the public domain, and to reflect the fact that the work had input from a wider team.
2. A decision-support system to diagnose factors limiting stream trout fisheries

2.1. Abstract

Maintaining or restoring productive freshwater fisheries is a key challenge for resource managers. However, the inherent uncertainty and complexity of managing fisheries, often based on scant environmental data, makes it difficult for managers and the public to reach consensus on appropriate actions. To help deal with this issue, we created a literature-based decision support system to diagnose limiting factors for stream brown trout fisheries. Once limiting factors are determined, appropriate management actions can be tailored to address them. Our Bayesian Belief Network-based framework (BBN) serves two functions: 1) it directs users to assemble a parsimonious environmental data set to inform stream fishery management, and 2) it integrates and interrogates these data to generate standardised and testable hypotheses about which environment factors are likely to limit trout productivity. The BBN has been trained on brown trout because among freshwater fish this species has the richest literature base and is highly valued worldwide. However, the framework could be adapted for other stream fish. We applied our BBN to the Horokiri Stream, a data-rich catchment in Wellington, New Zealand. The BBN probability outputs were comparable with the conclusions of five experienced fishery biologists following their detailed investigation into the factors that led to the loss of the Horokiri brown trout fishery between 1951 and 1990.

2.2. Introduction

Improving fisheries is a common motivation for stream protection and rehabilitation initiatives (Beechie et al. 2008). Yet effective fishery management is dependent on identifying and ranking the importance of environmental constraints on fish populations (Armstrong et al. 2003; Lake et al. 2007). If managers fail to identify limiting factors before acting they risk ecological
surprises (King 1995) or misdirecting resources (Beechie et al. 2008; Roni et al. 2002).

Underpinning the concept of limiting factors analyses (LFA) in fisheries biology is the Liebig-Sprengel ‘law of the minimum’ (Figure 2) – where yield is proportional to the amount of the most limiting factor (Liebig 1852; Sprengel 1839). For example, improving salmonid spawning habitat, by adding spawning gravels, in a stream that lacks sufficient adult fish cover will not result in a better fishery – because additional recruits created through this action will encounter a population bottleneck later in their life history. More recent ecological theory states that environmental pressures can interact in complex ways to affect populations (Townsend et al. 2008). In addition, populations can experience ‘co-limitation’ with multiple environmental factors (Sperfeld et al. 2011). For example, two stressors (e.g. reduced food availability and elevated water temperatures) can act synergistically to negatively affect fish growth or survival (Bruder et al. 2017). In this situation, alleviation of either stress on the population will result in increased density or biomass.

Figure 2. The Wurtsbaugh et al. (2015) version of a Liebig-Sprengel barrel for freshwater fish populations showing variables that can limit fish production (figure reproduced with lead author permission).
Determining the environmental factors that limit highly mobile, long-lived, upper-trophic level fish, such as brown trout (*Salmo trutta*), is challenging and resource-intensive. Consequently, managers are often faced with considerable uncertainty when assessing the potential outcomes of rehabilitation actions based on scant environmental data (Bash & Ryan 2002). To avoid inaction in the face of uncertainty, pragmatic managers will assign management resources based largely on common sense or intuition (Walters 2007). This issue is especially acute for small stream fisheries because investment of management resources is often unjustifiable given the relatively low fisheries value. Nevertheless, decision support tools are needed even for small-stream fisheries because collectively they can receive considerable angler usage across management jurisdictions, and can contribute recruits to main-stem fisheries.

In this paper we present and trial a Bayesian Belief Network-based (BBN) LFA framework for lowland stream brown trout populations. Bayesian Belief Networks are now widely used to support environmental decisions (Aguilera et al. 2011; Landuyt et al. 2013; O’Brien et al. 2018). For example, they have been used to predict the outcomes of different basin-scale fishery management plans, segment-scale river rehabilitation actions and land-use effects over broad spatial scales (Marcot et al. 2001; Borsuk et al. 2006; Quinn et al. 2013; Death et al. 2015). Our BBN differs from previous fishery management BBNs because by design it has relatively low data requirements. It is intended to provide a steady state ‘snapshot’ of segment scale trout habitat and population conditions to determine likely limiting factors and indicate the need and scope for further monitoring or management actions. This model needs to be applied repeatedly in order to capture dynamic processes (e.g. every 5 years). The BBN formalises the process of generating, integrating and interrogating a parsimonious data set for undertaking a LFA on small stream brown trout fisheries. To our knowledge, this is the first time that BBN modeling has been applied in this context.

We tested our BBN on the Horokiri Stream (formally known as Horokiwi Stream), a data-rich catchment in Wellington, New Zealand. To do this, we entered existing
environmental and brown trout population data into our BBN from before and after the well-documented decline of the stream’s trout fishery (between 1951 and 1990). The BBN outputs were compared to a narrative by five experienced fishery biologists who discussed the likely reasons for the population decline after a detailed investigation of the stream (Jellyman et al. 2000).

2.3. Methods

Model structure

We used Netica modeling software (version 5.23, Norsyes.com) to construct the BBN. Broadly, our network structure was modeled on medical diagnostic BBN-based decision support systems (Lucas et al. 2004). Initially, we created an influence diagram to conceptualise cause and effect linkages between key environmental variables and stream fishery health. Key limiting factors for stream fisheries, which were determined from our appraisal of the literature, were then extracted from the linkage diagram to populate a Directed Acyclic Graph (DAG). Limiting factor nodes were phrased as a positive statement (e.g. ‘Too Hot’ or ‘fish cover limited’) and placed within the middle of the DAG. We then populated the DAG with causal environmental variable nodes (parent nodes) using the initial linkage diagram as a guide. The basic structure of the BBN is shown in Figure 3. Subsequently, we undertook a broad review of the international salmonid literature and relevant technical reports to inform the decisions inherent within the BBN (see Appendix 1). The structure of the DAG constrained the scope of our review, although refinement of the DAG and further review of the literature was an iterative process.
Figure 3. The basic structure of our BBN (left) and example (right). In the top row are the parent nodes, which contain environmental variable information. In the next row are causally linked nodes conditional on the parent nodes. Below these are horizontally listed limiting-factor probability nodes. In the bottom row are the diagnostic test nodes, which contain fishery population metrics. An example of a simplified sub-net for ‘Food Limitation’ is shown on the right.

To simplify the BBN, where possible, we chose empirical data as input information rather than modeled proxies. For example, we require quantitative macroinvertebrate community composition data as an input variable in the Food-Limited sub-net branch. We did not attempt to model macroinvertebrate community composition based on causally linked variables - such as nitrate concentrations or deposited fine sediment. Our choice of parent node input variables was also filtered by environmental data that are practically obtainable, or potentially already exist, because they are extensively collected by water resource managers.

Determining parent node environmental variable categories
Following the approach of Marcot et al. (2006), we discretised environmental variables into broad categories within each parent node. For a given
environmental variable, our working definition of ‘not limiting’ was based on values that naturally occur in productive wild stream fisheries. We did not define the breakpoints for parent node categories relative to theoretical optima, which may not occur in natural streams. For example, continuous optimal water temperatures for salmonid metabolism occur only in controlled environments (e.g. hatcheries).

The environmental variable (parent node) category breakpoints were informed by either 1) linear interpolation, 2) literature-derived values in combination with author opinion, or 3) visual or quantile classification of data-distribution gradients (the former undertaken when breakpoints were obvious). For the latter method, we used two trout population and environmental spatial data sets from New Zealand. The first was from what is known as the ‘100 Rivers Study’, a nationwide multidisciplinary study in which trout abundance in 88 clear-water rivers was surveyed by snorkel divers (Jowett 1990, 1992). The second was a recent unpublished electrofishing study of 48 wadeable streams across a gradient of agricultural land-use intensity. This survey was undertaken as part of the Cumulative Effects Research Programme (CEP) Fishery Quality Study (C01X1005). The methods used to define the various category breakpoints are listed in Table 2 (Appendix 1, Section 1.1).

Limiting factor nodes

A conditional probability matrix is the functional link between BBN child nodes and parent nodes (Marcot et al. 2006). When determining values for conditional probability tables (conditional probabilities), we first weighted the relative ‘strength of influence’ of the parent nodes based on our literature review and/or author opinion. For example, the ‘flood-limited’ parent node variables ‘flood frequency’, ‘segment slope’ and ‘fish cover’ were weighted 1, 1 and 0.5, respectively. In this instance, fish cover was down-weighted because we suggest the presence of structural cover (from floods) in a stream is less important than
the occurrence and severity of large floods when determining a population response. For each parent node variable category, we assigned a standardised weighted score according to its negative, neutral or positive influence on the child node. For example, -4, -2, -1, 0 were used to score the four ‘flood frequency’ variable categories. Conditional probabilities (e.g. for flood limitation) were then calculated by summing all combinations of the parent node influence-weighted scores. The results were subsequently normalised to a 0–100 scale to represent the probability of the limiting factor child node’s logic statement being true.

In some instances, when supported by evidence, we accounted for potential synergistic or antagonistic interactions between parent node variables. For example, high temperatures are known to have a synergistic effect on dissolved oxygen (DO) stress in fish. Rather than adding influence-weighted standardised scores for the temperature and DO parent nodes, we multiplied the two if temperature and oxygen variable categories were above thresholds known to induce stress in brown trout.

**Diagnostic fish population metric nodes**

Generally, we used equal weighting for all parent nodes to determine the conditional probability table values that link the limiting-factor (parent) nodes with the diagnostic fish population metric (child) nodes. However, for the fish population metric nodes ‘trout biomass >200 mm’ and ‘young-of-the-year (YoY) density’, we developed intuitive conceptual models to estimate the cumulative effect of multiple limiting factors, or stressors, that are acting on a population (Figure 4). The values from these conceptual models (on the y-axis) were used to populate the conditional probability tables. The conceptual models are based on the principal that a proportion of a population will be resilient to a single stressor but the population will decline logistically as more stressors are added (Liess et al. 2016). For example, our conceptual model assumes that a population with one acting limiting factor will have a 90% chance of having a high large-trout biomass, whereas, when two limiting factors are in effect (e.g. flood limited and food limited) there is a 75% chance of intermediate biomass and 25% chance of high or
low biomass. Three (or more) acting limiting factors will result in a very high chance of low biomass (Figure 4A). Breakpoints for young-of-the-year density and biomass of trout >200 mm categories were determined with quantile classification of the CEP data.
Figure 4. Conceptual models used to determine conditional probability table values that link fish population metric nodes to limiting-factor nodes. The models define, depending on the numbers of cumulative limiting factors, (A) the likelihood of the occurrence of high, intermediate or low biomass of trout $>200$ mm, and (B) the occurrence of high, intermediate or low young-of-the-year trout densities (fish/m²). All probabilities sum to 100.
Model application

We assessed our BBN using historical data from Horokiri Stream – recorded in Allen (1951) and more recently in Jellyman et al. (2000). The Horokiri brown trout fishery collapsed between 1951 and 1990, and Jellyman et al. (2000) presents a comprehensive expert-based analysis of potential causes.

We entered the ecological and trout population data from Horokiri Stream from its 1990 ‘impacted’ state into our LFA BBN. These data were supplemented with six years of flow and water temperature data, and monthly water quality spot measurements (2002–2008), from the Wellington Regional Council long-term monitoring site (mid-catchment, easting 1761804, northing 5450652). We could not find suitable reference-state temperature, flow and forage-fish data, so we supplemented these nodes with present-day (impact state) data. We compared the limiting-factor probability outputs with Jellyman et al.’s (2000) narrative about the causes of the trout population decline. It is important to note that our BBN was not trained on the Horokiri Stream data. Therefore, this exercise represented an independent assessment of the BBN’s ability to generate limiting factor hypotheses (in the form of ranked probabilities) against the opinion of five experienced fishery scientists – who were informed by a targeted data collection exercise.

2.4. Results

Literature review

The literature review underpinning our BBN is provided in Appendix 1 and in Holmes et al. (2017). In these documents we give the rationale for including or excluding each potential limiting factor, suggest data requirements to populate the BBN parent nodes and cite the literature used to define breakpoints for parent node categories. The full BBN network structure is shown below in Figure 5.
Figure 5. The Bayesian Belief Network (version 5.23, Norsyes.com) directed acyclic graph for undertaking a limiting-factor analysis on stream brown trout fisheries. Limiting-factor nodes are located in the middle of the network, below the environmental parent nodes and above a set of fishery-population metrics (diagnostic nodes). Netica files available on request from the lead author.
Application of the limiting factor analysis

The narrative in Jellyman et al. (2000) suggests the decline of the Horokiri fishery was due to the stream becoming wider and shallower, in combination with reduced riffle area, undercut banks, and residual pool depths. These changes reduced macroinvertebrate production and the fish cover. In addition, increased benthic algae altered the macroinvertebrate community to one dominated by low food-value taxa (relative to the pre-1951 community). Jellyman et al. (2000) contend that the trout population became severely limited by food and cover. Allen (1951) comprehensively studied the Horokiri fishery prior to its decline, and co-authored the Jellyman et al. (2000) report. In the latter he suggests that the decline was due to “the apparent dramatic reduction in pool depth and undercut bank cover”.

Based on the Jellyman et al. (2000) electric fishing data, we estimated that post-decline YoY trout densities were 0.06 trout/m$^2$ and biomass of trout >200 mm was 0.22 g/m$^2$. This contrasts with the exceptionally high total trout biomass of 26.5 g/m$^2$ reported by Allen (1951) for the fishery in its reference state.

Based on post-decline environmental data alone (from Jellyman et al. 2000), the BBN ranked a lack of cover as the most likely limiting factor (81% chance of occurrence), followed by low-flow, recruitment and food limitation (Table 1). Limiting factors were ranked similarly when the BBN was re-run with the inclusion of the Jellyman et al. (2000) trout population data (Table 1). Based on Allen’s (1951) pre-decline environmental data, the BBN identified low-flows as the most likely limiting factor (50% chance of occurrence). After adding the reference-state trout population data, recruitment limitation became the most limiting factor (18 %) (Table 1).
Table 1. Bayesian belief network percent probability outputs for potential limiting factors in the Horokiri Stream brown trout fishery in its pre-1951 ‘reference state’ and recent ‘impacted state’ (based on data from 2000). Two sets of predictions are given for both states: 1) probabilities based on environmental data alone (i.e. leaving fishery population metrics undefined), and 2) probabilities based on environmental and fish population data.

<table>
<thead>
<tr>
<th>Limiting factor</th>
<th>Probability (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Reference state</td>
</tr>
<tr>
<td>Cover limited</td>
<td>44</td>
</tr>
<tr>
<td>Low flow limited</td>
<td>50</td>
</tr>
<tr>
<td>Recruitment limited</td>
<td>38</td>
</tr>
<tr>
<td>Food limited</td>
<td>37</td>
</tr>
<tr>
<td>Too hot</td>
<td>20</td>
</tr>
<tr>
<td>Flood limited</td>
<td>35</td>
</tr>
<tr>
<td>Water quality limited</td>
<td>6</td>
</tr>
<tr>
<td>Socially limited</td>
<td>0</td>
</tr>
</tbody>
</table>

Based on post-decline environmental data, the BBN predicted a 37% chance of low YoY trout density (<0.1 trout/m²) and a 20% chance of moderate density (0.1-1 trout/m²). The model also predicted a high (79%) probability of biomass of trout >200 mm being below 0.4 g/m². The model predicted that prior to 1951 (reference state) there was a 54% chance of YoY density exceeding >1 fish/m² (the highest density category), whereas there was a low (16%) chance of the biomass of trout >200 mm exceeding 4 g/m².

Using all available data, the relative changes in limiting-factor probabilities from pre- to post decline (i.e. impacted-state minus reference-state probabilities) were as follows: fish cover, low-flow, recruitment and food limitation increased by 80%, 70%, 47% and 35%, respectively. There was <25% change in all other limiting-factor probabilities.
2.5. Discussion

The BBN

Our model is underpinned by a substantial review of salmonid literature and relevant technical reports (Holmes et al. 2017). By consolidating this literature within a BBN, we created a novel and systematic framework for undertaking a Limiting Factor Analysis (LFA). The BBN approach allowed different types of information to be blended in a transparent manner. The result was a flexible weight-of-evidence approach to assess factors potentially limiting a fishery.

Our approach has advantages over previous expert assessment-based LFA. The LFA-BBN output probabilities indicate the severity of limitation by key environmental factors, and when ranked these probabilities specify the order in which limiting factors should be mitigated. Applying the BBN modeling process required limited input from fishery specialists. In addition, LFA-BBNs produce standardised numerical outputs, which change in a consistent way with varying environmental parameters. This ought to substantially reduce subjectivity inherent within repeat expert appraisals of potential fishery problems.

Our BBN is targeted at the segment-scale (i.e. 1-km stream segments). We anticipate the assessment could be scaled up by applying the process to multiple stream segments distributed within a catchment using a stratified randomised approach (e.g. Stevens & Olsen 2004). Used this way, the BBN could indicate how limiting factors vary in intensity within a catchment, allowing spatial targeting of appropriate management actions.

Creating our BBN was as much about deciding what to exclude from the modeling process as to what to include. Our goal was to create a tool to aid managers and community groups that are resource limited. Consequently, we have attempted to strike a balance between precision and practicality. Users are required to collect what we consider to be a parsimonious environmental data
set for undertaking a meaningful LFA to underpin stream and fishery management. We did not include an exhaustive list of limiting factors within the BBN.

We omitted some potential limiting factors because they will be self-evident or occur relatively infrequently. These include abiotic factors such as metal pollution or discrete catastrophic events (e.g. chemical spills). We also excluded two biotic factors - harvest and predation. With respect to harvest as a potential limiting factor, angling usage data (such as collected by New Zealand’s National Angler Survey; e.g. Unwin 2016) will provide some indication of fishing pressure. If fish abundance is relatively low in the face of substantial fishing pressure, and other potential limiting factors are considered unlikely (e.g. <25 % probability), then overfishing could be considered the primary limiting factor through a process of elimination.

In New Zealand’s lowland streams, cormorants, longfin eels and adult trout are the main potential natural trout predators (Hayes & Hill 2005; McDowall 1994). Despite the ubiquitous presence of these predators, New Zealand lowland fisheries have remained productive by global standards for over a century. Furthermore, no management options are available to ameliorate the effects of natural predators, other than adding structural fish cover, which we account for within the ‘fish cover limited’ BBN sub-net. Therefore, we excluded predation as a limiting factor within our model and argue that predation must be accepted within a fishery as a natural self-limiting constraint (Berryman 1992), much like the occurrence of temperatures below optima for trout metabolism.

**BBN application**

The initial application of our BBN shows that it can generate results that are consistent with expert opinion in degraded stream fisheries. The probability outputs were comparable with the expert narrative in Jellyman et al. (2000), explaining the decline in the Horikiri Stream brown trout fishery. They suggested that, post decline, the Horokiri trout population was severely food and cover limited. Likewise, our BBN suggests that the Horokiri fishery is probably limited
by a lack of cover (87% probability) and habitat availability at low flow (80% probability) - primarily because of scarce deep water at base flow. Food limitation was also ranked highly (43%), fifth among the eight potential limiting factors. In addition, based on environmental data alone, the BBN predicted a 79% chance of biomass of trout >200 mm being <0.4 g/m\(^2\) (the lowest biomass category). The observed biomass of trout >200 mm estimated from the Jellyman et al. (2000) data fell in the middle of this range (c. 0.22 g/m\(^2\)).

The BBN performed less convincingly when predicting limiting factors for the Horokiri’s reference state (pre-1951) – although some key environmental data were missing. Allen (1951) determined that the trout population was food-limited at this time, whereas the BBN gave just an 8% chance of food limitation and ranked recruitment limitation highest (18% probability). The inconsistency of the BBN outputs with Allen’s findings are largely due to our working definition of ‘food limitation’. Where Allen framed food limitation in terms of production, we define food limitation relative to food levels present in New Zealand stream fisheries with high trout biomasses. Pre-decline, the Horokiri Stream supported a very productive fishery. Moreover, trout biomass apparently exceeded the productive capacity of the invertebrate food base - leading to the ‘Allen Paradox’ which was a source of a lengthy scientific debate (Huryn 1996). Therefore, based on our definition, the low chance of food limitation given by the BBN is consistent with the high trout biomass observed in the stream pre-1951.

Interestingly, a subsequent investigation by Allen (1985) showed that Horokiri trout appeared to exceed maximum expected growth rates based on the stream’s temperature regime and assuming maximum invertebrate rations. This suggests that invertebrate food limitation may not have been as strong a constraint on productivity as originally assumed by Allen (1951).

Based on the reference state environmental data, the BBN correctly predicted a high chance of high juvenile trout density (54%) but incorrectly predicted a relatively low chance (16%) of high biomass of trout >200 mm. We did not have a
complete environmental data set for Horokiri Stream before 1951. Compatible temperature, flow and pelagic forage-fish abundance estimates were not available. These environmental variables may have been more favorable for trout production at the time. Nevertheless, based on the above results, we suggest that our BBN will provide more accurate limiting-factor probability outputs when applied to degraded streams.

The weakest component of our BBN is the subjective expert assessment of recruitment potential within the ‘recruitment-limited’ sub-net. We suggest that a BBN at least as complex as ours would be required to model recruitment limitation effectively. There are guidelines provided in Armstrong et al. (2003) for assessing the adequacy of spawning and juvenile rearing habitat within a catchment. We recommend that these are followed. However, if there is little information available to assess the recruitment capacity of a stream, we suggest this node should be held neutral (at ‘OK’) to reduce its potential to influence probability calculations within the BBN (as we did for Horikiri Stream). Developing a literature-based BBN to determine trout recruitment potential in stream catchments would be a useful way to consolidate the vast body of research available on this subject.

Despite its limitations, we have demonstrated that our BBN can generate sensible and objective hypotheses about limiting factors for trout in streams. Importantly, it does this with pragmatic data requirements to improve the management of data-poor fisheries. More broadly, our BBN is useful globally because it provides a flexible LFA template that can be tailored for any stream fish species. Adapting this template to other fish will make existing research directly applicable to species management and expose fundamental knowledge gaps.
3. Mechanically reshaping stream banks alters fish community composition

3.1. Abstract

Mechanically reshaping stream banks is a common practice to mitigate bank erosion in streams that have been extensively channelised and lowered for land drainage. A common perception regarding this activity is that fish populations will be largely unaffected, at least in the short term, because the low-flow wetted channel remains undisturbed. However, the response of fish populations to this practice has rarely been quantitatively evaluated. Using a Before-After-Control-Impact design, we assessed fish community responses to a catchment-scale bank reshaping event in a 4th-order low-gradient stream that drains an intensive agricultural landscape. Quantitative electric fishing and fish habitat data were collected two months before and annually for three years after the reshaping event. After reshaping, deposited fine sediment levels increased in impact reaches and there was a significant reduction in anguillid eel biomass (by 49%). In contrast, densities of obligate benthic gobiid bully species increased significantly in impact reaches—potentially due to reduced predation pressure from eels. Three years after bank reshaping, fish community structure had largely returned to its pre-impact state in the reshaped areas. Our results suggest that, even in highly modified stream channels, further bank modification can reduce instream habitat quality and displace eels for at least one year. Managers should endeavor to use bank erosion control measures that conserve bank-edge cover, especially in streams with populations of anguillid eels, because these fish are declining globally.
3.2. Introduction

Throughout recorded history stream channels have been modified to accommodate the spread and intensification of agriculture (Pierce et al. 2012). Removal of riparian vegetation, stream bed lowering and channelisation are among the most extensive modifications undertaken to improve land drainage and flood control (Brown 1974; Schoof 1980; Mattingly et al. 1993). These actions simplify stream environments with a range of ecosystem effects (see Blann et al. 2009 for a review), including reduced fish diversity (Gorman & Karr 1978). Channelised rivers also tend to support lower total fish biomass than less modified reference streams (Duvel et al. 1976, Frothingham et al. 2013).

In addition to its ecological effects, channelisation often leads to increased bank erosion (Blann et al. 2009). This occurs because the residence period of water in channelised stream segments is reduced—increasing peak discharge and the erosive power of floods (Mosley 1998; Bukaveckas 2007, Pedersen 2009). Moreover, if a stream bed is lowered it becomes more prone to undercutting and mass bank failure (Blann et al. 2009; Frothingham et al. 2013). The resulting elevated fine sediment loads from increased bank erosion can severely stress downstream ecosystems (e.g. Wood & Armitage 1999; Sutherland et al. 2002).

Mechanically reshaping stream banks (reshaping) is a common management practice to mitigate bank erosion in channelised and/or lowered agricultural streams (Hudson & Harding 2004; Simon & Rinaldi 2007; Kroger et al. 2015). Typically, this practice involves using diggers to reduce the cross-sectional angle of banks from the wetted edge (at base-flow) to bank-full. This results in lower-gradient banks that are less erosion-prone. Reshaping differs from channelisation because, if done carefully, the stream bed remains largely undisturbed and channel planform is unchanged. Yet, in common with channelisation, the stream edges are essentially denuded of vegetation and undercut banks are removed, reducing the amount of bank-edge and instream cover. Bank-edge cover is particularly important for large fish such as trout and anguillid eels (Glova 1999; Allouche 2002; Jowett et al. 2009).
Despite the conspicuous visual effects of reshaping on stream banks, there has been little research to quantitatively determine how fish populations are affected by the practice. The studies cited above focused on the effects of channelisation, not on reshaping previously channelised streams. Consequently, it is difficult for managers to balance the potential erosion-control benefits of reshaping with its instream ecological consequences. To address this knowledge deficit, we used a Before-After-Control-Impact (BACI) design (Stewart-Oaten & Bence 2001) to assess the response of a lowland-stream fish community to a large-scale reshaping event in a New Zealand stream. Within the reshaped stream segments, we predicted that bank-edge fish cover would be reduced and, as a result, large-bodied fish with a strong affinity for security cover features, such as eels, would decrease in abundance. Conversely, we predicted that populations of small, obligate benthic fish, such as gobiiids, would be unaffected by reshaping. This is because direct mechanical disturbance of the stream bed is typically avoided during reshaping and small benthic fish use patch-scale habitat features present within the streambed.

3.3. Methods

Study stream

We undertook our study in Waituna Creek, a 4th order stream in Southland, New Zealand (46°32′03″S 168°32′06″E). Over the past century, the stream has been channelised and lowered for most of its length to facilitate land drainage for farming. As a result, it has deeply incised banks—up to 3 m in places. Waituna Creek has an annual median discharge of 1.22 m³/s, is shallow (i.e. wadeable) and slow-flowing. Macrophyte growths, primarily introduced pondweed (*Potamogeton* spp.) and sweet grass (*Glyceria fluitans*), are ubiquitous throughout the lower reaches during early summer to late autumn (Tompson & Ryder 2007).
Stream-bank reshaping

A large-scale bank reshaping project was undertaken in Waituna Creek over the austral summers of 2014 and 2015. This project was initiated by Environment Southland (regional authority) in response to concerns that conspicuous bank erosion was causing elevated sedimentation rates in the downstream Waituna Lagoon, a large coastal wetland with exceptional cultural and conservation values (Tompson & Ryder 2007). The reshaping reduced the angle of both stream banks to a 1:2 slope over about 14 kilometres of the main stem. In most stream segments, the overall slope of the bank (averaged from the wetted edge to bank full) remained relatively unchanged or moderately lessened. However, steps in the bank batter and vertical undercut sections, created by localized erosion, were smoothed in an attempt to prevent bank slumping (Figure 6) (Draft Waituna Plan 2015).

Figure 6. Waituna Creek (mid catchment) looking upstream. Recently completed stream bank reshaping can be seen along the true right bank compared to a typical lower catchment bank profile (prior to reshaping) on the true left.
Study design
In 2014, three 40-m ‘impact’ reaches were selected within a 1 km segment of creek scheduled for reshaping. Three 40-m upstream ‘control’ reaches were selected within a 2 km segment where reshaping would not occur. Within the impact and control segments, sampling reaches were selected to have mesohabitat sequences, bank morphology and instream cover levels representative of the wider stream segment. All sampling reaches were separated by at least 60 m. The control segment was located approximately 4 km upstream of the uppermost impact study reach, between two large continuous segments of reshaped stream (Figure 7). These unavoidable study design features meant that replicate reaches within the impact and control segments were not truly independent. Further, the control reaches had potential to be impacted by effects of the reshaping that occurred in the wider catchment, for example by migration of fish or downstream drift of fine sediment from reshaped segments upstream.
Figure 7. The mid-lower Waituna Creek catchment (Southland, New Zealand) showing locations of impact and control reaches and the extent of the bank reshaping works.
The stream channel in both control and impact segments had been channelised and lowered historically (>50 years previously). The control segment had not been mechanically modified for at least 10 years prior to the first sampling event in 2014, as a result of the landowners not wanting the section of stream within their property being disturbed (pers. comm. S. Crump, Department of Conservation ranger, Southland). In contrast, prior to reshaping, the impact segment and stream segments in the wider catchment upstream of the control site had been subject to channel maintenance activities on a three- to five-yearly cycle. Maintenance activities included macrophyte clearing and riparian vegetation removal through herbicide spraying and mechanical removal. Accordingly, some pre-existing differences were apparent between impact and control reaches, with the latter having slightly steeper banks, a narrower channel and more instream cover and riparian vegetation. Other than the reshaping, no maintenance activities occurred in the impact or control segments during study period. The catchment-scale reshaping project was completed prior to the 2015 sampling event.

At all six reaches, instream habitat and fish populations were sampled two months prior to channel reshaping in March 2014 and then annually during late March from 2015–2017. On all occasions, sampling was undertaken over a period of five days during base-flow conditions following two weeks of stable flows.

Habitat data

Instream habitat data were collected at each study reach before undertaking electric fishing surveys. Habitat data were collected according to the instream component of the broad-scale stream habitat mapping protocol of Holmes & Hayes (2011). This protocol involves collecting percentage area data for habitat attributes including mesohabitat type (riffle, run, pool), water depth, instream cover, fine sediment cover (particle size category <0.2 mm, according to Wentworth 1922), and benthic algal cover (according to ‘nuisance’ filamentous and algal mat criteria in Biggs & Kilroy 2000). Depths and areas of instream cover
(mid-channel macrophytes and woody debris) and bank-edge cover (draping overhanging vegetation, emergent macrophytes and undercut banks) were determined using a combination of instream measurements and bankside visual estimates. For more details about the habitat sampling method, the field protocol is provided in the Appendix 2. Deposited fine sediment trapped in the upper streambed at each reach was assessed using the ‘shuffle test’ (Clapcott et al. 2011). This method involves assigning a qualitative score (1-5) to the sediment plume that results when the substratum is disturbed by vigorously twisting your boot in the stream bed. To reduce subjectivity inherent within visual estimate-based assessments, all habitat data were collected by the same experienced observer (lead author) on all sampling occasions.

**Fish population sampling**

At each study reach, stop-nets (6-mm mesh) were simultaneously placed at the upstream and downstream boundaries and secured to the substratum. Fish populations within stop-netted reaches were sampled by electric fishing using the multiple depletion-pass method (Johnson et al. 2007). Three or more passes were undertaken until densities of eels, trout or bullies were <25% of the first pass. Two Smith Root (LR24) back-pack electric fishing machines (each fished with one anode, in tandem) were used to systematically fish the entire reach in a downstream direction. The depletion-pass number of all fish were recorded, as were lengths and weights—unless fish were too abundant. Diadromous gobiiids (commonly called bullies in New Zealand) (*Gobiomorphus* spp.) and īnanga (*Galaxias maculatus*) were occasionally very abundant. In these cases, 50–100 fish were randomly selected from the first pass and weighed, measured and identified to determine a site-specific average weight and, for the bullies, a species ratio. Following this, fish were weighed in batches and average weight and species ratio were used to determine species abundances from the total batch weights. The total wetted area of stream between the stop-nets was measured to allow converting fish data into densities and biomass per square m.
Data processing

Total abundance was estimated from depletion counts using the maximum weighted-likelihood approach (Carle & Strub 1978). We could not achieve depletion for inanga or lamprey (Geotria australis). For these species, aggregate numbers from all passes were used to determine abundance.

For most species, an index of biomass was determined by the total weight of all fish from the first three passes. We undertook the following calculations to estimate total biomass (g/m²) of longfin eels (Anguilla dieffenbachii) and brown trout (Salmo trutta) for each reach. First, we subtracted the total number of fish caught (from all passes) from the depletion count-based total population estimate. This gave the estimated number of fish in the reach additional to the fish that were caught. We then multiplied the number of additional fish by the geometric mean weight of all fish sampled from within the reach. This figure was added to the combined weight of fish from a reach, giving a total biomass estimate which was subsequently divided by the reach area.

To determine the responses of habitat and fish metrics to the bank-reshaping event, for each sampling reach we subtracted the metric values calculated from data obtained during the 2015, 2016 and 2017 surveys (after reshaping) from the metric values obtained during the 2014 survey (before reshaping). This gave a set of three before-after comparisons for both the control and impact segments for each ‘after’ sampling occasion. Variance (SE) was determined for each set of three before-after comparisons for the control and impact segments.

Statistical analysis

We used repeated-measures ANOVAs (rmANOVA) computed in SPSS 20 (IBM SPSS Statistics; IBM Company, Chicago, IL, U.S.A) to determine the relative changes in fish and habitat metrics for each BACI comparison. In this analysis, the between-subjects factor was ‘reach type’ (control versus impact reaches) and ‘time’ (the
four sampling occasions) was the within-subjects (repeated-measures) factor. The key outputs from this analysis were three specific contrasts for the ‘time*reach type’ interaction: 1) ‘before reshaping’ (the first sampling date) versus ‘one year after’ reshaping, 2) before versus ‘two years after’ and 3) before vs. ‘three years after’. If one or more of these contrasts were significant, this indicated an effect of the bank reshaping treatment. A significant ‘between-subjects’ effect indicated that there was an overall difference (averaged across all four sampling dates) between control and impact reaches. Such overall differences could be due to pre-existing differences between the two reach types because the sampling occasions included one ‘before’ and three ‘after’ dates. Of lesser interest is a significant ‘within-subjects’ effect of ‘time’ indicating that a response variable has changed over time but this change is unrelated to bank reshaping. The significance level for all tests was set at $P=0.10$ due to the small sample size (just three replicates per reach type) and low overall statistical power (Quinn & Keough 2002). For each result, effect sizes (partial eta squared values, range 0–1) are presented to allow an evaluation of their likely biological relevance (Nakagawa & Cuthill 2007; effect size categories: very weak <0.10, weak >0.10, moderate >0.30, strong >0.50). All response patterns described below were significant ($P<0.10$) unless otherwise indicated.

To investigate potential relationships between fish response variables and continuous habitat metrics of interest (e.g. longfin eel biomass and bank-edge cover), we also pooled data from all reaches for linear regression analysis. These were run for each combination of variables for each year to determine if the strength of the apparent relationships changed over time. Only significant responses ($P<0.10$) are presented.

### 3.4. Results

Over the four sampling occasions, eleven fish species were found in total. In order of overall total abundance (summed across all sites and sampling occasions), these included: common bullies (*Gobiomorphus cotidianus*) $n=17,042$, inanga (*Galaxias maculatus*) $n=3,379$, redfin bullies (*Gobiomorphus huttoni*) $n=2,321$, giant bullies...
(Gobiomorphus gobioides) n=645, longfin eels n=432, lamprey (Geotria australis) n=141, brown trout (Salmo trutta) n=99, black flounder (Rhombosolea retiaria) n=12, shortfin eels (Anguilla australis) n=8, giant kōkopu (Galaxias argenteus) n=3 and kōaro (Galaxias brevipinnis) n=1. Summed across all sites and sampling occasions, common bullies comprised 42% of the total biomass, with longfin eels, trout and īnanga contributing approximately 30%, 13% and 6%, respectively. All other fish species made up less than 5% of the total biomass.

The results of the rmANOVAs are given in Table 2. For brevity, only response variables that showed a significant ‘time*reach interaction’ or ‘between-subjects’ effect are presented. All significant results had ‘strong’ effect sizes based on their partial eta squared values, indicating they are most likely biologically relevant (Nakagawa & Cuthill 2007). When interpreting these results, the relative changes between impact and control reaches need to be considered. For instance, if fish biomass in impact reaches increased after reshaping but this increase was smaller relative to the increases in control reaches during the same year, this indicates a negative effect of reshaping on fish biomass.
Summary of repeated-measures ANOVAs (P-values and partial eta² effect sizes) comparing habitat and fish species metrics before and after the stream bank reshaping event. For within-subject effects, P-values and effect sizes are given for the specific contrasts. These include before vs. one-year-after (B/1yrA), before vs. two-years after (B/2yrA), and before vs. three-years-after (B/3yrA). Only variables with at least one significant between- or within-subjects response (at α = 0.10) are shown. Reach type abbreviations: C = Control; I = Impact. Effect size categories (after Nakagawa & Cuthill, 2007): very weak <0.10 (probably biologically irrelevant), weak >0.10, moderate >0.30, strong >0.50.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Between-subject effects</th>
<th></th>
<th></th>
<th>Within-subject effects (specific contrasts)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Reach type</td>
<td>P-value [effect size]</td>
<td>Reach type ranking</td>
<td>Time P-value [effect size]</td>
</tr>
<tr>
<td>Mean wetted width</td>
<td></td>
<td>0.04 [0.70]</td>
<td>C &lt;I</td>
<td>B/1yrA: 0.08 [0.58]</td>
</tr>
<tr>
<td>Bank-edge fish cover</td>
<td></td>
<td>&lt;0.01 [0.89]</td>
<td>C &gt;I</td>
<td>B/2yrA: 0.07 [0.60]</td>
</tr>
<tr>
<td>Fine sediment cover</td>
<td>0.89 [0.01]</td>
<td></td>
<td></td>
<td>B/1yrA: 0.05 [0.66]</td>
</tr>
<tr>
<td>Shuffle test score</td>
<td>0.88 [0.01]</td>
<td></td>
<td></td>
<td>B/1yrA: 0.07 [0.61]</td>
</tr>
<tr>
<td>Longfin eel biomass</td>
<td>0.15 [0.44]</td>
<td></td>
<td></td>
<td>B/1yrA: 0.03 [0.51]</td>
</tr>
<tr>
<td>Common bully density</td>
<td>0.21 [0.36]</td>
<td></td>
<td></td>
<td>B/1yrA: 0.09 [0.87]</td>
</tr>
<tr>
<td>Redfin bully density</td>
<td>0.71 [0.04]</td>
<td></td>
<td></td>
<td>B/1yrA: 0.07 [0.61]</td>
</tr>
<tr>
<td>Lamprey density</td>
<td>0.07 [0.61]</td>
<td></td>
<td>C &gt;I</td>
<td>B/1yrA: 0.08 [0.57]</td>
</tr>
<tr>
<td>Īnanga density</td>
<td>0.45 [0.15]</td>
<td></td>
<td></td>
<td>B/1yrA: 0.03 [0.71]</td>
</tr>
</tbody>
</table>

Habitat variable responses

Wetted stream width was significantly affected by bank reshaping, increasing in impact reaches relative to controls one and two years after reshaping (a respective mean increase of 17% and 10% relative to initial state; Table 2, Table 3, Figure 8a). One year after reshaping, bank-edge cover had decreased by an average of 40% in
impact reaches, whereas it had increased by 60% in control reaches (Figure 8b); however, these differences were not significant (P=0.14, eta2=0.30), potentially because of very high variation between replicate reaches. Two and three years after reshaping, bank-edge cover had increased in both impact and control reaches (Table 2, Table 3, Figure 8b).

Table 3. Mean values and relative percent change for variables from before reshaping to after reshaping for the three ‘after’ sampling occasions. Values in bold show where repeated-measures ANOVA revealed a significant (P <0.10) within-subjects time * reach interaction (which indicates an effect of the bank reshaping treatment) and/or a significant between-subjects effect (equivalent to an overall difference between impact and control reaches across all four sampling dates, which may indicate pre-existing differences between the two reach types). A negative percentage value indicates a decrease relative to before the bank reshaping event.

<table>
<thead>
<tr>
<th>Variable</th>
<th>Before mean Control values [and SE]</th>
<th>Before mean Impact values [and SE]</th>
<th>% change at Control reaches</th>
<th>% change at Impact reaches</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean wetted width (m)</td>
<td>4 [0.07]</td>
<td>6.3 [0.20]</td>
<td>-17</td>
<td>-10</td>
</tr>
<tr>
<td>Bank-edge fish cover m²/reach area</td>
<td>0.2 [0.04]</td>
<td>0.8 [0.01]</td>
<td>60</td>
<td>46</td>
</tr>
<tr>
<td>Shuffle test score</td>
<td>3.2 [0.33]</td>
<td>2.7 [0.33]</td>
<td>37</td>
<td>47</td>
</tr>
<tr>
<td>Longfin eel biomass (g/m²)</td>
<td>12.4 [3.2]</td>
<td>8.9 [1.1]</td>
<td>160</td>
<td>-9</td>
</tr>
<tr>
<td>Common bully density (no./m²)</td>
<td>0.57 [0.07]</td>
<td>0.30 [0.06]</td>
<td>1138</td>
<td>534</td>
</tr>
<tr>
<td>Redfin bully density (no./m²)</td>
<td>0.07 [0.03]</td>
<td>0.07 [0.03]</td>
<td>2264</td>
<td>245</td>
</tr>
<tr>
<td>Lamprey density (no./m²)</td>
<td>0.06 [0.03]</td>
<td>0.001 [0.00]</td>
<td>20</td>
<td>187</td>
</tr>
<tr>
<td>Ūnanga density (no./m²)</td>
<td>7.89 [1.80]</td>
<td>8.11 [4.72]</td>
<td>-100</td>
<td>-97</td>
</tr>
</tbody>
</table>
Figure 8. Mean change (after–before) in habitat variables for the three sampling occasions after bank reshaping: (a) mean wetted width, (b) bank-edge cover, (c) fine sediment cover, and (d) shuffle test scores. Error bars show SE. Control reaches are represented by solid bars and impact reaches by hashed bars (n = 3).
Both measures of instream deposited fine sediment were affected by bank reshaping (Table 2, Table 3, Figure 8c,d). Relative to before reshaping, fine sediment cover on the bed increased in impact reaches by 31% one year after reshaping, whereas it decreased in control reaches by 35%. Two years after reshaping, these respective differences between control and impact reaches relative to their initial state remained similar. For all three years following reshaping, shuffle-test scores increased relative to those estimated before reshaping at both control and impact reaches. One year after reshaping, however, shuffle-test scores had increased more strongly in impact reaches (by 75%) than in control reaches (37%). Percentage macrophyte cover, mesohabitat diversity, percentage bank-edge cover, average depths and percentage algal cover did not change significantly from before to after reshaping.

**Fish responses**

Longfin eel biomass, common bully density and redfin bully density were all affected by bank reshaping (Table 2, Table 3, Figure 9). One year after reshaping, eel biomass had increased overall compared to before reshaping but was reduced in impact reaches (by 49% compared to before) whereas a strong increase (by 160%) occurred in control reaches. Common bully densities had increased at both reach types in all three years following reshaping. However, one year after reshaping, common bully densities in impact reaches had increased far more than in control reaches (by 4409% versus 1138%). Redfin bully densities increased strongly at both control and impact reaches in the first year after reshaping.
Figure 9. Mean change (after–before) in longfin eel biomass (a), redfin bully density (b), and common bully density (c) for the three sampling occasions after bank reshaping. Error bars show SE. Control reaches are represented by solid bars and impact reaches by hashed bars (n = 3).
Overall, lamprey densities were higher in control than in impact reaches, but no time-by-treatment interactions occurred for this species, indicating that this difference between reach types existed prior to reshaping (Table 2, Table 3). Giant bullies showed no significant responses to bank reshaping. Black flounder, shortfin eels, giant kōkopu and kōaro were too rare to warrant inclusion in the statistical analysis. Wide variability in brown trout abundance prevented a meaningful statistical analysis with respect to this species.

**Relationships between longfin eel biomass and bank-edge cover**

Before bank reshaping occurred, there was a positive (but insignificant) relationship between longfin eel biomass and bank-edge fish cover across all six study reaches (Figure 10). One year after reshaping, this relationship became significant (P=0.02) and the slope of the relationship became steeper. By three years after, the relationship had disappeared and longfin eel biomass was unrelated to the amount of bank-edge cover in reaches.
Figure 10. Relationships between bank-edge cover (m²/m² of stream reach area) and longfin eel biomass for before bank reshaping, one year after, two years after and three years after reshaping. Data were log transformed (base 10). The R and P values (from linear regression analysis) for each relationship are given in the top left corner of each plot.

3.5. Discussion

Few studies have experimentally evaluated the response of freshwater fish communities to mechanically altering stream banks. Most existing studies are correlative in nature or assessed fish population response to habitat improvements (Bernhardt et al. 2005, Smokorowski & Pratt 2007). There are cover removal studies which could be considered analogous to our study. However,
these studies focused on waterways with fish communities dominated by salmonids (e.g. Chapman & Knudsen 1980, Dolloff 1984) or are limited to reach-scale habitat manipulations (Jowett et al. 2009). We quantitatively evaluated the response of a primarily obligate benthic stream fish population to a catchment-scale channel engineering event. To our knowledge, this is the first experimental investigation of this kind.

In our study, resource constraints limited replication and a lack of analogous catchments meant that control reaches had to be situated within the same stream and downstream of a long, reshaped segment. In addition, the sample reach replicates within the control and impact segments were not truly independent, because each reach was broadly representative of habitat and fish populations within its wider segment. Finally, there were pre-existing differences between impact and control sites, with control sites tending to be narrower and with more instream cover because the control stream segment had been subjected to less channel maintenance in recent years. In a perfect BACI design, there would be no pre-existing differences between independent sites, and control reaches would be isolated from any treatment effects (Stewart-Oaten & Bence 2001). Below we interpret our findings keeping in mind the unavoidable limitations of our study design.

In line with our first hypothesis that large-bodied fish species with a strong affinity for cover features would be negatively affected by bank reshaping, longfin eel biomass was significantly reduced in impact reaches relative to controls one year after reshaping. Conversely, by one year after reshaping, there was a much stronger increase in common bully densities in impact compared to control reaches (Table 3, Figure 9c). This latter result did not support our second hypothesis, which had predicted that populations of small, obligate benthic fish species such as common bullies would be unaffected by bank reshaping.
Instream structural cover and longfin eels

There was a conspicuous reduction in bank-edge cover (i.e. overhanging and emergent vegetation and undercut banks) in the impact reaches. Overall, bank-edge cover had reduced by 40% in impact reaches when assessed one year after reshaping, whereas it had increased by 60% in control reaches. This was largely because of the growth of bank-edge emergent macrophytes (R. Holmes, pers. obs.). Although the changes to bank-edge cover were not statistically significant (due to very high variation between the few reach type replicates), the moderate effect size ($\eta^2=0.30$) suggests this change was nonetheless biologically relevant (see Nakagawa & Cuthill 2007). By two years after reshaping, bank-edge cover had increased at both impact and control sites, again primarily due to growth of emergent macrophytes.

Large longfin eels are commonly associated with instream structural cover (Glova & Sagar 1994; Broad et al. 2002, Glover et al. 2010). Moreover, instream cover is thought to be a common limiting factor for longfin eel biomass in streams (McDowall 1990; Glova et al. 2010). For example, Jowett et al. (2009) showed that removing bank-edge cover from a stream reach reduced longfin eel biomass by half. In effect, the present study is a catchment-scale replication of Jowett et al.’s small-scale cover removal experiment. Accordingly, a reduction in longfin eel biomass at the impact sites following reshaping was expected, although we were surprised by the magnitude of the effect. Longfin eel biomass halved in impact reaches whilst increasing three-fold in control reaches one year after reshaping (Table 3, Figure 9a). The disproportionally large increase in eel biomass in control sites, relative to the more moderate decrease overserved in impact sites, may reflect the proportionately large area of stream that was affected by reshaping in the wider catchment (Figure 7).

Some regrowth of bank-edge vegetation had occurred by the first ‘after’ sampling occasion, approximately one year after reshaping. However, immediately following reshaping, the stream segment containing the control reaches was effectively an island of ‘cover-rich’ habitat (about 3 km long) within a stream
where the banks had been completely denuded of edge cover for several
kilometres, both up- and downstream (Figure 6, Figure 7). Consequently, we
suggest the substantial increase in longfin eel biomass in control sites was a result
of longfin eels migrating there from the wider catchment to seek out the structural
cover. In support of this interpretation is the emergence, and subsequent
disappearance, of a strong correlation between eel biomass and bank-edge cover
across all sites over the three years following bank reshaping (Figure 10). It
appears that bank-edge cover was limiting longfin eel carrying capacity in impact
reaches in the first two years after bank reshaping. Three years after reshaping,
following the regrowth of bank-edge vegetation, cover became abundant enough
to no longer limit eel biomass in impact reaches. By this time, the longfin eel
population appears to largely resemble its pre-impact distribution (Figure 9a).
With only one ‘before’ sampling period we cannot discount that the initial
sampling period represents an anomalous year in terms of eel biomass for
comparison with subsequent years. However, the observed response of longfin
eels to the reshaping fits with relevant theory. Moreover, the return of the eel
population to pre-reshaping biomass distribution three years later suggests that
our initial sampling period was representative of typical fish community
conditions.

Small benthic fish species
By one year after bank reshaping, abundance of common bullies had increased by
4000% in impact reaches. This increase was far greater than the corresponding
increase in control reaches. We suggest that the relative increase in common
bullies at the impact sites was an indirect result of the bank reshaping through a
reduction in abundance of large longfin eels. Large eels are the top predator in
Waituna Creek and bullies are a common prey item (McDowall 1990; Jellyman
2010). Similarly, some authors have speculated that because habitat removal can
disproportionality affect large predatory fish, this can negate the effect of reduced
Instream deposited fine sediment

One and two years after bank reshaping, percent fine sediment cover on the streambed surface in impact reaches increased, in contrast with a strong, sustained decrease in control reaches during the same period (Table 3, Figure 8c). Similarly, shuffle-test scores, which are an indication of the amount of fine sediment trapped in the top 10 cm of streambed, consistently increased more strongly in impact than in control reaches over the three years following reshaping, especially in the first year (Figure 8d). These findings indicate that bank reshaping caused a substantial amount of fine sediment to enter the stream and settle on, and within, the stream bed. This result is in keeping with personal observations made by the lead author. One year after reshaping, the survey team found it difficult to work in the impact reaches because of patchy deep mud (10–30 cm) deposits, which were largely absent prior to reshaping. It is important to note these results were observed despite reshaping occurring upstream of the control sites, meaning that some downstream drift of sediment into the control sites from the river works upstream would be expected. The increase in deposited fine sediment in impact reaches may have contributed to the observed changes in the fish community. Longfin eels are thought to prefer stream habitat with low levels of deposited fine sediment (McDowall 1990).

The fine sediment increases observed after reshaping are concerning because the management justification for the bank reshaping works was to reduce fine sediment entering the ecologically sensitive Waituna Lagoon downstream (Draft Plan for Waituna 2015). Our results suggest that large-scale bank reshaping can increase fine sediment loading in low-gradient streams. Moreover, within the control segment, the stream banks appeared stable for the duration of the study, calling into question the need for channel maintenance activities to be undertaken across large spatial scales or over routine sub-decadal cycles.
Conclusions

Our BACI assessment revealed that a major stream channel reshaping event had significant instream physical and ecological consequences—despite a high chance of reporting false negative results because of low sample replication. We observed an increase in instream deposited fine sediment and an apparent redistribution of a substantial portion of the longfin eel population to a stream segment that was unaffected by the reshaping. Reshaping also appears to have affected the stream food web. One year after reshaping, common bullies underwent a population explosion at the impact sites—in parallel with a reduction in the biomass of predatory longfin eels.

Three years after the reshaping event, the impact sites largely resembled their pre-impact state based on the habitat and fish population metrics investigated. However, it must be reemphasized that, prior to the reshaping event, the impact sites had undergone intensive channel engineering (i.e. channel straightening and bed lowering), as well as regular (every 3-5 years) macrophyte-clearing operations (Greer et al. 2017). Both these practices are known to considerably disturb stream communities (e.g. Chapman & Knudsen 1980, Kaenel et al. 1998, Smokorowski & Pratt 2007). Consequently, our study provides a conservative example of the ecological effects of streambank reshaping. The observed effects are likely to be more severe and/or longer-lasting if the practice is applied to relatively unmodified stream channels. We also show that reshaping counterproductively increased instream fine sediment levels, at least temporarily, highlighting a need for managers to evaluate the efficacy of reshaping as a bank erosion control measure. It is beyond the scope of the present study to determine if the apparent short-term increase in deposited fine sediment will be offset by a long-term reduction in fine sediment supply as a result of the reshaped banks. We suggest using alternative techniques such as two-stage channels, which reduce bank erosion while conserving fish cover (Powell et al. 2007), especially in streams with
populations of anguillid eels because these fish are declining globally (Jacoby et al. 2015).
4. Riparian management affects instream habitat condition in a dairy stream catchment

4.1. Abstract

Using a space-for-time substitution design, we investigated the response of structural instream habitat and fish populations to different riparian management practices throughout a Dairy Best Practice Catchment. We found a significant negative correlation between the upstream area of stock exclusion fencing and deposited instream fine sediment cover. Furthermore, we determined that this relationship emerges when ≥ 300 m lengths of upstream riparian area were included in the analysis - indicating the scale at which stock exclusion fencing results in a positive instream habitat response. Specifically, for this historically degraded spring-fed stream, our findings indicate that riparian segments with 5-m wide stock exclusion fences (both banks) are required to achieve instream fine sediment cover below 20% in downstream reaches. Fish were sparse and evenly spread throughout the catchment. Fish distributions were not correlated with reach-scale riparian or instream habitat variables, possibly because the available habitat quality gradient was too narrow.

4.2. Introduction

One hundred and fifty years of agricultural development have resulted in degraded water quality and macroinvertebrate communities in the majority of New Zealand’s lowland streams (Quinn et al. 1997; Quinn 2000; Larned et al. 2004). There is perception amongst the public and fishery managers that freshwater fishery values are also declining as a result of land-use intensification (Jellyman et al. 2003). This perception is fuelled, in part, by conspicuous and ongoing changes to the structural habitat of farmland streams through channelisation and the removal of riparian vegetation. Longfin (Anguilla
dieffenbachii) and shortfin (A. australis) eels and the introduced brown trout (Salmo trutta) support highly valued customary, commercial and recreational fisheries in New Zealand (McDowall 1994; McDowall 2011). The impact of land-use induced structural habitat changes on these stream fisheries remains unquantified but is considered to be substantial (Jellyman et al. 2003; Beentjes et al. 2005).

A key structural habitat change in streams with agricultural catchments is an increase in the deposited fine sediment load (Allan 2004; Matthaei et al. 2006). This is caused by run-off from agricultural land, reduced bank stability as a result of vegetation clearance and livestock activity within a stream network (Trimble & Mendel 1995; Wood & Armitage 1999; Lyons et al. 2000). Fine sediment homogenises stream habitat by reducing residual-pool depth and smothering the stream bed. In turn, this reduces fish refugia during periods of low flow and production of benthic invertebrates, a key food source for fish (Duncan & Ward 1985; Wood & Armitage 1997; Allouche 2002; Sutherland et al. 2002; Gayraud 2002; Korstrom 2006). These changes can ultimately reduce the carrying capacity of a stream for fish (Waters 1995; Bjornn et al. 1977; Richardson & Jowett 2002; Mossop & Bradford 2006). Furthermore, fine sediment fills interstitial spaces within coarse substrata, which provide cover for juvenile eels and brown trout (Raleigh et al. 1986; Glova et al. 1998; Glova 2002; Jowett & Richardson 2008). In addition, deposited fine sediment in salmonid spawning redds slows hyporheic exchange reducing oxygen levels and metabolite flushing and consequently egg survival rates (Sternecker & Geist 2010).

Recently, there has been an exponential increase worldwide in resources targeted at rehabilitating stream habitats degraded by agricultural or urban development (Roni et al. 2002; Lester & Boulton 2008; Stewart et al. 2009; Bernhardt et al. 2011). Stream fishery rehabilitation initiatives are based on the premise that fish (and their prey) will respond positively to increased structural habitat diversity and improved water quality (see “field of dreams hypothesis”, Palmer et al. 1997). In New Zealand, excluding livestock from the stream edge and re-establishing
riparian vegetation underpins most rehabilitation projects (Parkyn et al. 2003; Dodd et al. 2009; Greenwood et al. 2012; Wahl et al. 2013). The potential benefits of instating ungrazed riparian marginal strips are considerable and include:

1) establishment of dense ground cover vegetation and more porous soils that filter fine sediment and particulate-bound contaminants from overland flow, 
2) reduced nitrogen and phosphorus loading through degassing and assimilation into plant matter, 
3) reduced stream bank erosion through the establishment of protective vegetative ground cover and soil-binding root masses, 
4) reductions in stream temperatures and nuisance algal and macrophyte growths through shading, 
5) increased allochthonous resources through the supply of terrestrial insects and leaf matter, and 
6) increased cover for fish through draping vegetation and the supply of debris to streams (Platts & Wagstaff 1984; Quinn et al. 1997; Parkyn et al. 2003; Niyogi et al. 2007; Yuan et al. 2009; DeWalle 2010; Stewart-Koster et al. 2010; Zhang et al. 2010; Wahl et al. 2013).

Pastoral retirement has been shown to benefit some stream salmonid populations overseas (Keller & Burnham 1982; Summers et al. 2008). Conversely, in New Zealand, Jowett et al. (2009) found that removal of riparian vegetation from a well-vegetated stream caused a subsequent reduction of eel density and biomass. However, a meta-analysis and review of fishery rehabilitation projects (Roni et al. 2002) and several other reviews of the international riparian rehabilitation literature conclude that the responses of biota to rehabilitation efforts are often underwhelming (Kondolf & Micheli 1995; Bash & Ryan 2002; Parkyn et al. 2003; Hillman & Brierley 2005; Lester & Boulton 2008; Stewart et al. 2009). Typical reasons for fish failing to respond positively to stream rehabilitation include:
1) rehabilitation efforts are often limited to the reach-scale and thus fail to address overriding stream-segment or catchment-scale disturbances,

2) lag periods for habitat or fish population responses to rehabilitation actions can exceed practical monitoring periods, and

3) factors other than habitat quality (e.g. dispersal or fishing) may limit the ability of biota to respond to localised habitat improvements.

Given the inconsistent performance of rehabilitation projects there remains a need for further research to inform effective riparian enhancement measures, particularly in relation to valued fisheries. Specifically, identifying the scales at which riparian management actions result in improved fish habitat and fish populations is a key research question.

The Dairy Best Practice Catchment (DBPC) Programme undertaken in New Zealand from 2001 to 2011 provided an opportunity to investigate long-term ecosystem responses to catchment-scale stream rehabilitation. It involved applying best on-farm management practices (on a voluntary basis) throughout five degraded lowland wadeable streams across New Zealand (Waikakahi, Inchbonnie, Bog Burn, Toenepi and Waiokura). Physicochemical water quality was intensively monitored over ten years. The key findings of the programme were that stock exclusion fencing and improved farm effluent management reduced suspended sediment and total phosphorus levels, whereas nitrate levels remained constant or increased (Wilcock et al. 1999; Wilcock et al. 2007; Monaghan et al. 2009). Nevertheless, despite stakeholder workshops identifying brown trout fisheries as a key value in three of the DBPC’s (Waikakahi, Inchbonnie and Bog Burn), the responses of fish habitat and fish populations to the rehabilitation measures were not investigated.

The present study was undertaken in the Waikakahi Stream, one of the DBPCs. Our two research aims were to establish whether riparian habitat variables correlate with structural instream fish habitat quality and fish populations, and to
determine the longitudinal scale at which relationships between riparian management practices and high quality instream fish habitat emerge. Because landowners in the Waikakahi catchment have invested varying degrees of effort in managing the riparian area on their properties, this effectively provides a gradient of riparian habitat quality within the catchment. We collected instream and riparian structural habitat data from the stream using a catchment-scale fish habitat survey protocol, and also quantitative eel and trout population data from eight stream reaches spanning the entire gradient of habitat quality available within the catchment. Based on the related previous research cited above, we developed three specific hypotheses:

1) Instream habitat in reaches within, or immediately downstream of, an established (i.e. in place for two or more years) and relatively wide (> 5 m on each bank) fenced riparian management area (henceforth called ‘fenced riparian area’) will have greater mesohabitat diversity, more stable banks, decreased widths (through encroachment of stream edge vegetation), reduced deposited fine sediment levels, greater depths and increased amounts of instream fish cover relative to reaches with small or poorly maintained riparian management areas;

2) The positive relationship between the size of fenced riparian area and instream fish habitat quality indicators will become stronger as the length of fenced riparian area upstream of ‘response’ reaches increases;

3) Trout and eel density and biomass will be related to the proportion of high quality instream habitat (i.e. relatively deep, structurally diverse habitat with high proportions of fish cover and low levels of deposited sediment).
4.3. Methods

Study stream
The Waikakahi Stream is a fifth-order, spring-fed tributary of the Waitaki River located on the coastal plains of South Canterbury (Figure 11). It is approximately 18 kilometres long with a catchment area of 63 km². Annual rainfall is between 600 and 700 mm (Monaghan et al. 2009). An irrigation development in the 1980s has caused the flow regime to be augmented by the nearby Waitaki River. Irrigation ‘wipe-off’ water from extensive border-dyke (flood) irrigation has resulted in mean summer base-flows 10 times higher than those recorded prior to the irrigation development. Near its confluence with the Waitaki River the stream has a median flow of 0.57 m³ s⁻¹ (range = 0.028 – 3.180 m³ s⁻¹). The stream catchment is now dominated by dairy farming (>90% of catchment land area) following conversion from sheep and beef cattle during the early to mid-1990s (Monaghan et al. 2009).

Study design
Riparian and structural instream habitat in the Waikakahi Stream were surveyed during base-flow conditions between 27 February and 1 March 2012 using a broad-scale habitat mapping protocol detailed in Holmes & Hayes (2011) and Holmes et al. (2016). The survey included a desktop analysis of existing catchment knowledge (as described in Harding et al. 2009) to identify strata that dictate the selection of survey sites. According to this protocol, stratification was not necessary in the Waikakahi because catchment-scale influences were similar throughout the stream’s length (i.e. the whole catchment was relatively low gradient throughout with no major tributaries or land-use changes).

Eight c. 1 km stream survey segments (Figure 11) were randomly chosen from a pool of 17 potential segments within the main stem of Waikakahi Stream. Potential segments were selected using a generalised random tessellation stratified (GRTS) spatially balanced survey design (Stevens & Olsen 2004).
Figure 11. The Waikakahi Stream catchment (South Canterbury, New Zealand) showing the locations of the eight surveyed 1 km stream segments. The stream flows from west to east before its joins the Waitaki River c. 4k upstream of the coast. All tributaries and the mainstem 1k upstream of Site 8 are ephemeral.

Riparian survey

At each site, a two-person field team ground-truthed riparian habitat features on a set of four to six continuous ortho-rectified aerial photographs of the entire survey segment (1:700 scale). Imagery was sourced from Environment Canterbury http://gis.ecan.govt.nz/arcgis/services. Information was recorded directly onto the aerial photographs and included the lengths and widths of the fenced riparian area, broad categories of riparian vegetation (based on successional stage from grasses to forest), location and areas of trees, land use type (e.g. dairy) and land use features (e.g. stock crossings), and potential contaminant sources (e.g. bank slumping and stock pugging). For an example of the field instructions and a completed section of the riparian survey see Appendix 3.
**Instream survey**

Three 100 m reaches, each split into five continuous 20-m sub-reaches, were randomly nested within each of the eight 1 km survey segments (115 sub-reaches in total, 5 reaches were inaccessible). Within each sub-reach, various habitat attributes were recorded as visual estimates (as m/m² or a percentage of the sub-reach). Habitat attributes included meso-habitat types (riffle, run and pool), depths (categories 0-0.3, 0.3-0.5, 0.5-1 and >1 m), cover attributes (macrophytes, overhanging vegetation, undercut banks and woody debris), substratum characteristics including particle size categories according to Wentworth (1922), and the occurrence of ‘nuisance’ filamentous and mat forming algae according to criteria in Biggs & Kilroy (2000). Definitions of recorded instream habitat features, and an example of the instream survey field sheet are provided in Appendix 2. To gather coarse water velocity information a representative fast-run, medium-run and slow-run were chosen from the lower Waikakahi Stream, and mean velocity in each was estimated by recording the time taken for a tennis ball to drift a known distance. Each estimate was the mean of three drift trials. A coefficient of 0.85 was used to convert surface velocity to mean channel velocity (US EPA 2015).

**Fish population survey**

Eight 40-m reaches were systematically selected to capture the entire gradient of instream habitat quality available for adult trout and eels within the surveyed areas of the catchment. In relative terms, ‘low habitat quality’ reaches were shallow and wide, with high levels of deposited fine sediment and low amounts of instream cover. ‘High habitat quality’ reaches were relatively deep with low deposited sediment and high amounts of instream cover.

Fish population surveys were undertaken during base flows on 28 November 2012 and again on 15 March 2013. Fish populations were estimated for each stop net bounded 40-m reach using multiple depletion passes with two Smith Root (LR24) back-pack electric fishing machines. Captured fish were anesthetised with AQUI-
S® (a clove-oil based fish anesthetic). Once ataxia was observed, weights and lengths were recorded. All fish were released near their capture sites. Owing to low fish numbers overall and consequent difficulty in achieving depleting counts, aggregate fish numbers from all passes were used for abundance analysis. Instream habitat was re-surveyed at each fishing reach during each fish sampling occasion.

**GIS analysis**

Riparian habitat data were digitised by scanning and ortho-rectifying the ground-truthed aerial photographs within ArcView GIS version 10.1. All habitat information was then transferred to GIS by creating shape files of the hand-drawn information. Information about each habitat feature was transferred to attribute tables for each shape file. The eight 1 km survey segments were divided into 100 m longitudinal ‘GIS-defined Riparian Zones’ (GDZ), each with a 30-m-wide area either side of the stream to encompass all riparian habitat features. All GDZ had small differences in area depending on the sinuosity of the channel (i.e. a GDZ would be slightly smaller if it included tight bends in the stream channel, or larger if it included areas at the extreme end of a survey segment and thus gained extra length). Each GDZ had an area of approximately 6000 m² (range 5900-8800 m²).

**Statistical analyses**

Within each GDZ, riparian data were converted to rasta to enable the area of each feature to be used as inputs for regression modeling. All instream habitat data were converted to total areas or mean percentage area cover of the wetted area for each 100 m reach to standardise the measurements. A riparian vegetation index was calculated by weighting vegetation category types according to their successional stage from pasture to mature multi-storied canopies. Category weightings were as follows: mixed exotic grass (× 1), mixed native and exotic grasses (× 2), mixed native and/or exotic shrubs and grasses (× 3), swampy and/or emergent mixed native and exotic grasses and herbs (× 4), mature trees with
understory vegetation (× 5). Vegetation category scores were then applied to each square metre of riparian area within a GDZ, these scores were then summed to give an area weighted riparian vegetation index score. A maximum score of 60,000 would be achieved by a GDZ containing 100% mature forest (i.e. total GDZ area × 5).

Statistical analysis was undertaken using R statistical software. After exploratory data analysis, percentage (p) data (e.g. cover) were logit-transformed as follows to normalise the data, a requirement of the parametric statistical tests used:

\[
\text{logit}(p) = \log \left( \frac{p}{100 - p} \right)
\]

This transformation emphasises differences near the extremes (0% and 100%) and deemphasises differences near the centre (50%). Other data (counts and right-skewed) were log-transformed to meet the assumption of normality.

After matching instream habitat survey reaches with their corresponding GDZ, exploratory analyses were undertaken by constructing linear regressions (with linear, quadratic and cubic terms) for all measured riparian predictor variables against each instream variable. This process was first undertaken using the mean of all data available for a given riparian variable at each survey segment regressed against the combined mean for an instream habitat variable from all three instream survey reaches within the survey segment. In this analysis, we assumed that the three instream survey reaches, which were randomly nested within each riparian survey segment, were representative of instream habitat within the entire segment.

To investigate the effect of scale on potential riparian and instream habitat relationships we repeated the riparian vs. instream regression analysis including only the mean values for a given riparian variable measurement for 500 m upstream of each of the instream reaches (i.e. the values from five continuous GDZ). For this analysis our sample size was reduced because some instream
Habitat reaches were positioned less than 500 m downstream of the upstream end of a riparian survey segment. These reaches were excluded from the analysis. A third ‘reach scale’ regression analysis was undertaken comparing the instream habitat variables with riparian variables in the immediately adjacent GDZ.

To investigate the possibility that significant relationships were being masked by other predictor variables, we conducted multiple linear regressions using the full set of available predictors and a backward model selection process. No further significant relationships were revealed. The small sample size prevented investigation of interaction terms.

The relationship between instream deposited fine sediment and the area of fenced riparian area was examined further by including all data from the instream survey reaches. The observed instream sediment cover levels at a given reach could be the cumulative result of variation in riparian habitat predictors upstream. To account for this, we weighted the response variable (% fine sediment cover) according to the position of the instream survey reach within the riparian survey segment. Instream reaches were weighted sequentially, with reaches located at the top of a survey segment weighted at one tenth of those located at the bottom of the segment. In addition, we also conducted a series of 10 sequenced regression analyses that incrementally included the mean fenced riparian area of contiguous GDZ upstream. Starting at 100 m (reach scale), the fenced riparian area from the adjacent GDZ is used as the predictor value. For each subsequent regression, the mean fenced riparian areas of upstream GDZ were incrementally included (i.e. ≤ 200 m upstream, ≤ 300 m upstream etc.). The instream habitat reaches were located randomly within the riparian survey segments. Therefore, sample size was reduced as more GDZ were incrementally included in subsequent regressions.

For fish population data, exploratory analysis was undertaken by constructing regression matrices of instream habitat and fish population variables for both the November and March sampling occasions. As with the riparian and instream
habitat data, multiple linear regressions using the full set of available instream habitat predictors were performed.

4.4. Results

Instream habitat survey

Overall, 2470 m of the instream habitat (c. 15% of the Waikakahi stream’s total length) and 8500 m of the riparian area (c. 50% of the stream’s length) in the mainstem was surveyed. Mean water velocities for ‘typical’ fast, medium and slow runs were estimated to be 0.6, 0.5 and 0.3 m/s, respectively. Water velocity for fast runs gives an indication of the fastest habitats in the stream (excluding shallow riffles). ‘Slow run’ was the dominant meso-habitat type, averaging 48% of the surveyed habitat (Table 4). The depth categories 0-0.3 m and 0.3-0.5 m combined comprised 70% of the surveyed area whereas habitat with water depths exceeding 1 m contributed only 6% (Table 4). Macrophyte beds were a dominant feature of the stream, with a mean cover of 38%. Coarse gravel (grain size 16-32 mm) was the most common substratum size category, comprising an estimated 39% of the streambed surface.

Deposited fine sediment cover was highest in the upper Waikakahi stream. The two uppermost survey segments had mean fine sediment covers of 63% and 36%, respectively, whereas the overall catchment mean was 19%. Fine sediment and fine gravel together made up 39% of the stream bed (Table 4). Instream cover was predominantly overhanging vegetation and emergent and submerged aquatic macrophytes (Table 4). Undercut banks were also reasonably common comprising 15% of the stream edge. Minor amounts of woody debris, submerged branches and man-made cover objects were present (Table 4).
Table 4. Mean percentage values (and standard deviations) for instream habitat variables, including mesohabitat type, depths and sediment cover, for all 20 m sub-reaches combined (n = 115). Streambed sediment categories were only for the visible areas of the stream bed (i.e. areas not obscured by deep water or aquatic macrophytes). Fish cover categories include Ub = Undercut bank, Ov = Overhanging vegetation, Lwd = Large woody debris, Sb = Submerged branches, Mm = Man-made cover). Ub and Ov are the mean % linear cover of the 20-m sub-reach margins (i.e. 40 m in total over both banks). Lwd, Sb and Mm are the mean % stream-bed cover of the wetted area of the 20 m sub-reaches.

<table>
<thead>
<tr>
<th>Meso-habitat types</th>
<th>Pool</th>
<th>Riffle</th>
<th>Slow run</th>
<th>Med. Run</th>
<th>Fast run</th>
<th>Backwater</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>2.7 (14.6)</td>
<td>4 (13.9)</td>
<td>48 (47.2)</td>
<td>14 (34)</td>
<td>31 (42.4)</td>
<td>0.4 (2.7)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Depths</th>
<th>0-0.3m</th>
<th>0.3-0.5m</th>
<th>0.5-1.0m</th>
<th>&gt;1m</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>29.3 (36.2)</td>
<td>40.5 (33.9)</td>
<td>24.7 (27)</td>
<td>5.9 (15.1)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Stream bed sediment cover</th>
<th>Clay/mud/silt/sand</th>
<th>Fine gravel</th>
<th>Coarse gravel</th>
<th>Small cobble</th>
<th>Large cobble</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>19.3 (29.5)</td>
<td>19.3 (19.8)</td>
<td>38.6 (28.4)</td>
<td>19.4 (21.6)</td>
<td>3.7 (8.1)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Fish Cover types</th>
<th>Ub 0-0.3m</th>
<th>Ub 0.3-0.5m</th>
<th>Ov 0-0.3m</th>
<th>Ov 0.3-0.5m</th>
<th>Ov 0.5-1m</th>
<th>Ov &gt;1m</th>
<th>Lwd m²</th>
<th>Sb m²</th>
<th>Mm m²</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>9.1 (20.2)</td>
<td>5.6 (16.5)</td>
<td>30.9 (38.8)</td>
<td>25.4 (37.5)</td>
<td>3.9 (15.8)</td>
<td>4.6 (17.3)</td>
<td>0.2 (1.1)</td>
<td>0.2 (3.2)</td>
<td>0.5 (0.32)</td>
</tr>
</tbody>
</table>
Riparian habitat

Grazed pasture dominated the areas of the ‘GIS-defined Riparian Zones’ (GDZ). Un-grazed exotic grasses were the predominant vegetation category within the fenced riparian areas, although minor amounts of other vegetation types were also present (Table 5). Exotic willows (*Salix* spp.), poplars (*Populus* spp.), pines (*Pinus* spp.) and macrocarpa (*Cupressus macrocarpa*) were the most common riparian tree species. Native plantings of trees and grasses associated with rehabilitation projects were present in <0.2% of the catchment’s riparian area. The distribution of riparian vegetation index scores was right skewed, confirming the dominance of rank grass as the most common vegetation type in the catchment. The mean riparian vegetation quality index score was 6840 (range 1990 – 9540).

Table 5. Mean percentages (and standard deviations) for the coverage of various riparian vegetation categories for all GIS-defined riparian zones (GDZ) n = 86. With the exception of wetland-type vegetation, the vegetation categories represent the successional stages from grazed pasture through to mature forest (a more detailed explanation of the vegetation category types can be found in Appendix 3).

<table>
<thead>
<tr>
<th>Riparian category</th>
<th>Grazed pasture</th>
<th>Exotic grasses</th>
<th>Native and exotic grasses</th>
<th>Native and exotic grasses, shrubs and trees</th>
<th>Swampy/emergent native and exotic grasses and herbs</th>
<th>Trees (without understory vegetation)</th>
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<tbody>
<tr>
<td></td>
<td>81.6 (12.5)</td>
<td>13.5 (12.3)</td>
<td>0.7 (2.11)</td>
<td>0.2 (1.0)</td>
<td>3.7 (6.82)</td>
<td>0.05 (0.5)</td>
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</table>

Effective stock exclusion fencing occurred in all but four of the 86 GDZ. Evidence of livestock activity (i.e. stock pugging) occurred in seven GDZ. In GDZ where stock pugging occurred, the mean length of pugged areas was 17.6 m / GDZ (range 9.9-32.2 m). Active bank slumping occurred in 23 GDZ; the mean length of slumping was 17.5 m / GDZ (range 3.2-42.5 m). In total, 331 m of bank slumping was recorded in the catchment – or 2% of the total length of surveyed stream bank. Overall the GDZ had a mean fenced riparian area of 1140 m² (range 0-3430 m²).
This equates to a mean fenced riparian margin width of 5.7 m on each bank (range 0-17.2 m).

Riparian and instream habitat correlations

Overall, 621 regressions were performed for all possible riparian and instream habitat combinations at three spatial scales (reach scale – 100 m, sub-segment scale – 500 m and segment scale 1000 m). Of these, 198 (or 32%) were statistically significant (P = <0.05). This percentage is far higher than the 31 (just under 5%) significant regressions one would expect to emerge by chance alone. Nevertheless, using a conservative approach, we report only those of primary interest (i.e. those that relate directly to our hypotheses (Table 6). Fenced riparian area was negatively correlated with deposited fine sediment at the sub-segment and segment scales but not at the reach scale. Fenced riparian area was negatively correlated with bank slumping at the reach scale. Fenced riparian area was also negatively correlated with recent stock access (as indicated by stock pugging) at the sub-segment and segment scale, but not at the reach scale. Stock pugging was positively correlated with the occurrence of instream fine sediment at all spatial scales (Table 6). Consistent with our hypotheses, the strength of the negative relationship between instream fine sediment cover and fenced riparian area increased as we increased the spatial scale of analysis (Table 6). Therefore, this relationship was chosen for more detailed analysis.
Table 6. Correlations from an exploratory regression analysis of all riparian and instream habitat variables. Regressions were undertaken at three scales: 100 m reach scale \( n = 23 \); 500-m sub-segment scale \( n = 15 \); and 1000-m segment scale \( n = 8 \). Only results that were significant \( (P < 0.05) \) for regressions that relate to our hypotheses are shown. Moderate \( (R^2 = 0.3 - 0.5) \) and weak \( (R^2 < 0.3) \) significant positive relationships are denoted by vertical and upwards sloping arrows respectively. Moderate \( (R^2 = 0.3 - 0.5) \) and weak \( (R^2 < 0.3) \) significant negative relationships are denoted by vertical (downwards pointing) and downwards sloping arrows respectively. A horizontal dash denotes no significant relationship.

<table>
<thead>
<tr>
<th>Habitat variable</th>
<th>Riparian habitat variable</th>
<th>Mean fenced area m^2 / GDZ</th>
<th>Mean stock pugging m / GDZ</th>
<th>Mean bank slumping m / GDZ</th>
<th>Mean riparian veg. index score / GDZ</th>
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<tr>
<td></td>
<td></td>
<td>100 m</td>
<td>500 m</td>
<td>1000 m</td>
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<td>Fine sediment (% cover)</td>
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<td>Bank Slumping (m/reach)</td>
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<td>Overhanging veg. 0.5-1m (% cover)</td>
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<td>Fine sediment (% cover)</td>
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<td>Undercut banks 0 - 0.3m % cover</td>
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<td>Depth 0 - 0.3m % of reach</td>
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<td>Depth 0.5 - 1m</td>
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<tr>
<td>Mesohabitat diversity</td>
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<td>Stock pugging (m/reach)</td>
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<td>Undercut banks 0.3 - 0.5m (% cover)</td>
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<td>Undercut banks 0 - 0.3m (% cover)</td>
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<tr>
<td>Depth 0.3 - 0.5m % of reach</td>
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<td>Depth &gt;1m % or reach</td>
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</table>

Instream fine sediment cover was significantly negatively correlated with the area of ‘fenced riparian area with full stock exclusion’ (henceforth called ‘fenced riparian area’). Furthermore, the strength of this relationship increased over the
three scales tested (1 km, 500 m, 100 m). The non-linear, negative shape of the relationship between fenced riparian area and deposited fine sediment at the stream segment scale is illustrated in Figure 12.

![Figure 12](image)

Figure 12. Regression between longitudinally weighted mean % fine sediment cover and mean fenced riparian area per 100 m of stream length (n = 23) for all GDZ within each stream segment (1000 m). Circle size represents the weight (0 through 10) applied to each % fine sediment response value according to the location of the 100 m reach within the 1 km survey segment. For example, fine sediment cover values from a reach at the top of a survey segments were weighed one tenth the value of a reaches at the bottom of the segment.

A significant negative relationship between fenced riparian area (m²) and instream fine sediment cover emerged when a 300 m section of upstream riparian area was included in the sequenced regression analysis (Figure 13). This relationship was strongest when a 400 m upstream riparian area was included, after which it
dropped below the P = 0.05 significance level at 600 m (probably because of reduced sample size). The non-linear, negative shape of the relationship at 400 m upstream riparian area, where it was strongest, is shown in Figure 14.

Figure 13. Summary of P, adjusted $R^2$ and sample size values for 10 individual (unweighted) sequential regressions of instream fine sediment cover versus mean fenced riparian area. The first regression, at the 100 m reach scale, compares sediment cover in instream survey reaches to fenced riparian area from adjacent GIS Defined Zones (GDZ). For each subsequent regression, the fenced riparian area is the mean from upstream GDZ incrementally included in the analysis.
Figure 14. Mean deposited fine sediment cover from 100 m instream reaches regressed against the mean fenced riparian area of four upstream continuous 100 m GDZ (n = 15). The relationship shown (the fourth regression in a sequence of 10 regressions that included predictor variables from upstream areas in 100 m increments) had the strongest $R^2$ value.

**Instream habitat and fish populations**

Trout and eels averaged 3-5 individuals (of each species) per 40 m electric fishing reach (range for all species: 0-11 individuals per reach). Trout densities averaged 0.02 no./m$^2$ in November (range 0.007-0.05) and 0.01/m$^2$ in March (range 0.003-0.03). Brown trout >200 mm outnumbered juvenile trout (<200 mm) 2:1 in November and 10:1 in March. Trout biomass averaged 8.5 g/m$^2$ in November (range 14.1-2.9) and 9.7 g/m$^2$ in March (range 23.9-1.0). Few juvenile eels (<400 mm) were captured on either sampling occasion (3 in total). Total eel densities averaged 0.02 no./m$^2$ during November (range 0.002-0.05) and 0.01 no./m$^2$ during March (range 0.0-0.02). Longfin eel biomass averaged 15.4 g/m$^2$ in November (range 0-39.7) and 10.2 g/m$^2$ in March (range 0-23.3 g/m$^2$). Shortfin eel biomass
averaged 9.7 g/m² in November (range 0-58.5) and 2.5 g/m² in March (range 0-15.8).

In November, longfin and shortfin eel densities were negatively related to wetted width (Figure 15A) whereas, trout biomass showed a positive linear relationship with overhanging vegetation (Figure 15B). In March, none of these relationships were significant but total trout biomass was positively related to wetted width (Figure 15C). There were no other significant relationships between fish population metrics and the measured habitat variables.
Figure 15. (A) Linear regression between mean density (no./m²) of longfin and shortfin eels (combined) and wetted width (of instream survey reaches) in November 2012 (B), mean trout biomass (g/m²) and overhanging (draping) vegetation expressed as mean % cover of the wetted width in November 2012; and (C) total trout biomass (g) and wetted width in March 2013.
4.5. Discussion

Excessive levels of suspended and deposited fine sediment were present throughout the Waikakahi Stream following dairy conversion in the 1980s (Meredith et al. 2003). In part, this provided the impetus to implement a catchment-scale fencing initiative. Riparian fencing and improvements to effluent management were suggested as the primary actions that led to a four-fold reduction in suspended fine sediment levels in the Waikakahi over the ten years of the Dairy Best Practice Catchment programme (Monaghan et al. 2009; Dodd et al. 2009). The present study shows that stock exclusion fencing, as part of a wider best practice programme, can provide significant benefits by increasing bank stability and maintaining relatively low levels of deposited fine sediment instream.

Fish habitat

We found a significant negative relationship between instream fine sediment cover and fenced riparian area in the Waikakahi catchment, supporting our first hypothesis that relatively wide fenced riparian areas would have a positive influence on various aspects of instream eel and trout structural habitat quality. Moreover, consistent with hypothesis two, the strength of this relationship emerged as significant as the spatial scale of analysis was increased. Sequenced regressions that included the fenced area upstream as a predictor variable, in increments of 100 m, showed the negative response of deposited fine sediment to fenced riparian area emerged as significant at ≥ 300 m (Figure 12, Figure 13). This suggests that the relatively large fenced riparian areas that are present along at least 300-m lengths of the Waikakahi stream maintain relatively low levels of deposited fine sediment downstream.

Decreased stock access, increased bank stability and increased filtering of overland flow are all potential mechanisms by which relatively large fenced riparian areas can reduce fine sediment entering a stream (Trimble & Mendel 1995; Wood & Armitage 1999; Lyons et al. 2000; Pusey & Arthington 2003). Indeed, relatively large fenced riparian areas appear to have a localised stabilising
effect on adjacent stream banks in the Waikakahi catchment. Fenced riparian area was negatively correlated with the occurrence of active bank slumping at the reach scale. In addition, relatively large riparian areas also appear to afford better stream bank protection from stock. Stock activity, as evidenced by pugging on the stream margins, was significantly negatively related to fenced riparian area at the sub-segment and segment scale but not at the reach scale – indicating that stock travelled short distances from riparian access points before accessing waterlogged soil. Unsurprisingly, recent stock pugging within the riparian management area was significantly positively related to deposited fine sediment cover in the adjacent stream reach. The effect strength of this relationship increased cumulatively with regressions that accounted for greater upstream areas (Table 6). These results agree with an established body of research linking stream edge stock activity and to sediment input in streams (e.g. Trimble & Mendel 1995).

Riparian vegetation can have a stabilising effect on stream banks, although in some instances larger trees have been shown to cause channel instability (Sweeney & Newbold 2014). In our study there was no apparent effect of riparian vegetation category type on bank stability at the various spatial scales investigated. However, the Waikakahi catchment lacks substantial areas of established dense riparian trees. Therefore, our analysis is limited to comparing fallow exotic grasses with short reaches of flax plantings or areas with sparsely planted exotic trees and (wilding) willows. All GDZ had low vegetation index scores (range 1990 – 9540) relative to the maximum possible score (60,000).

Fine sediment cover was highest in the upper Waikakahi Stream suggesting limited sediment transport capacity between survey segments. A similar pattern of relatively high deposited sediment levels in upper catchments of small Canterbury Plain drains and streams was observed by Greenwood et al. (2012). The bed-flushing power of the Waikakahi Stream is augmented by Waitaki River irrigation water (Meredith et al. 2003). This augmented flow probably complements the riparian fencing initiative by flushing stored sediment, the legacy of dairy conversion and subsequent poor farming practices, reducing chronic sediment
load and increasing stream flushing power via encroachment of stream-edge vegetation narrowing the channel (Greenwood et al. 2012). We cannot discount that flushing power, and hence sediment transport rates, vary moderately throughout the low gradient mainstem. However, channel form is similar over the entire sample frame and the channel widens to accommodate the moderately increased flows downstream. Meredith et al. (2003) reports that high deposited sediment levels were ubiquitous throughout the catchment following dairy conversion. This suggests that conditions conducive to fine sediment loading and deposition were likewise similar throughout the catchment prior to the stream fencing initiative.

**Fish populations**

Overall, and contrary to our third hypothesis, we did not find consistently higher fish densities or biomass in reaches with “high” habitat quality (i.e. relatively large amounts of fish cover, low amounts of deposited fine sediment and relatively deep water). Native eels and exotic brown trout were sparse and evenly distributed throughout the catchment on both sampling occasions, numbering just 0-11 individuals (of each species) per 40 m electric fishing reach. Possibly, there was too little differentiation between our high- and low-quality habitat reaches to influence fish distributions within the catchment. Alternatively, our reach-scale habitat and fish population survey was too coarse to detect any fish-habitat linkages at the low population densities observed. Densities of trout were an order of magnitude lower than those observed in streams that are approaching carrying capacity (Ayllón et al. 2010; Grant & Kramer 1990; Hayes 1988). Similarly, densities of both longfin and shortfin eels were very low compared with a survey of small streams from a range of land use types which had not been commercially fished (Hicks & McCaughan 1997). Even our ‘low habitat quality’ electric fishing reaches had patch scale (1-5 m²) cover features (e.g. macrophyte beds or small pockets of deep water) that could have provided adequate cover for one or two large eels. As with trout, the low densities of eels across all our sampling reaches probably meant that habitat was not saturated so we observed little discrimination in abundance over the habitat quality gradient. A third reason is that electric fishing
may have been more effective in the shallower, less structurally complex ‘low quality’ fishing sites. Deeper water and relatively large amounts of vegetative debris in the high-quality sites may have allowed fish to evade capture during all electric fishing passes (Hickey & Closs 2006).

Hay (2004) conducted a trout spawning survey in the Waikakahi Stream and suggested that the trout population may be recruitment limited. He observed high egg mortality rates and suggested this was as a result of excessive levels of deposited fine sediment and possibly high nitrate concentrations. In our survey, adult trout outnumbered juvenile trout in all reaches during both sampling occasions, which would be consistent with Hay’s hypothesis. Olsson & Persson (1988) found that deposited fine sediment cover levels >20% negatively affected trout embryo survival rates. Our survey revealed that deposited fine sediment levels in the Waikakahi catchment are generally above this threshold, especially in the upper catchment.

In the case of eels (which do not reproduce instream) there is no clear reason for the apparent low densities. The Waikakahi, a tributary of the Waitaki River, is a low-elevation catchment near the coast with low flow variability and no known passage barriers for elvers migrating upstream. However, based on elver catches at the base of the Waitaki Dam over the past decade, recruitment of longfin and shortfin eels in the Waitaki River system may be minimal in comparison with the available habitat (Jellyman 2012). In addition, our study could not account for the possibility of commercial fishing occurring. Eel fishers selectively target ‘high quality habitat’ in easily accessible lowland streams such as the Waikakahi. Furthermore, commercial fishing methods can substantially reduce eel biomass in a single night and population recovery may take up to 10 years if immigration is factored out (Chisnall et al. 2003; Jellyman & Graynoth 2005).

**Good practice riparian management**

The New Zealand dairy industry’s ‘Sustainable Dairying: Water Accord’ (2013) currently recommends that streams wider than 1 m should be fenced, but it provides no guidance on how wide riparian fencing should be. A recent review and
meta-analysis by Sweeney & Newbold (2014) concluded that riparian areas should be at least 30 m wide to protect key aspects of forested small stream ecosystems. Thirty-metre-wide forested riparian management areas are obviously unrealistic within intensively farmed pastoral landscapes. However, our findings suggest that some instream values, particularly those linked to deposited fine sediment (e.g. salmonid spawning and juvenile eel habitats), may already be protected by substantially narrower riparian areas.

Reducing or maintaining fine sediment cover below 20% could be an appropriate target to protect invertebrate food resources and recruitment of eels and trout in streams (Olsson & Persson 1988; Crisp & Carling 1989; Clapcott et al. 2011; Burdon et al. 2013). In our study, on average, deposited fine sediment levels were below 20% in 1 km stream segments that had 1000 m² of fenced riparian area per 100 stream metres (Figure 12). This area equates to a mean fenced width of five metres along each bank. We put this figure forward as an interim recommendation for wadeable spring-fed streams - with the caution that it is based on a correlative study in a single stream catchment and is unlikely to be directly transferable to rain-fed streams. We expect that wider fenced riparian areas would be required in streams with greater erosive power, although decreased channel stability, and thus increased sediment loading, might be countered by increased flushing of bed sediments.

Further research is needed to determine if riparian fencing widths, that adequately protect instream values, vary in proportion with catchment and segment-scale parameters (for instance, land use, flow source, stream geomorphology or disturbance regime). In anticipation of this research, our paper presents a survey and analysis method that can be applied on a catchment-by-catchment basis to indicate the riparian fencing widths required to reduce instream deposited fine sediment below a target value.
5. PHD SYNTHESIS AND DISCUSSION

Below I provide a synthesis of the preceding five chapters of this thesis before discussing the general implications of my research. Within each chapter summary, the outcomes of the thesis are interpreted with specific reference to contemporary freshwater management in New Zealand. Following the chapter summaries, I present a stream management ‘thought experiment’ to illustrate an interesting concept that emerges from my thesis. Finally, nested within this discussion, I present a conceptual model (intended for future publication as a ‘short communication’) that aims to provide a framework for incorporating spatiotemporal scale into future stream rehabilitation science and practice.

5.1. Chapter 1: Introduction

In my introduction chapter, I broadly summarise the river restoration/rehabilitation literature (hereafter: rehabilitation). Within the relevant literature, the lament that most rehabilitation projects fail to adequately assess ecosystem outcomes emerged as a repetitive theme (e.g. Bernhardt et al. 2005, Beechie et al. 2008). Individual project resources are often exhausted on operational aspects, with little or no reserves left for monitoring. Moreover, because ecological performance is very hard to quantify, the rehabilitation projects that do have some form of self-evaluation tend to focus on process-based measures to determine success, rather than performance or outcome-based measures (Hillman & Brierly 2005). Rehabilitation projects in New Zealand are no exception. For example, to date, rehabilitation success has been based almost exclusively on measures such as ‘kilometres of stream fenced’ or ‘number of riparian plants planted’. The notion that New Zealand’s rehabilitation initiatives, such as the Dairy Accords, are achieving their desired ecological outcomes is largely based on optimistic inference from relevant ecological theory.
There are a handful of projects that have assessed ecological outcomes, most notably the Dairy Best Practice programme conducted from 2001 to 2011 (Monaghan et al. 2009). However, these studies were largely limited to measuring water quality attributes (Collins et al. 2013). Very few New Zealand studies have attempted to experimentally define causal connections between large-scale stream habitat rehabilitation and fish community dynamics (see Jowett et al. 2009 for an exception).

Broadly, my thesis attempts to address this knowledge deficit and evaluate the ecological responses of instream physical habitat and fish to habitat manipulation and rehabilitation. My secondary objective was to develop practical science-based methods to help managers and stream-care groups restore stream fisheries. To this end, in Chapter 2, I review the salmonid literature to create a knowledge base to underpin a Bayesian Belief Network-based Limiting Factor Analysis (BBN-LFA) for stream trout fisheries. In Chapter 3, I develop and test the BBN-LFA on a stream fishery. In the subsequent two chapters, I quantitatively evaluate the fish community response to a catchment-scale mechanical stream-bank engineering event (Chapter 4) and assess the instream ecological outcomes of a catchment-scale stream rehabilitation initiative (Chapter 5).

5.2. Chapter 2. A decision-support system to diagnose factors limiting stream trout fisheries (Holmes et al. 2018)

In this chapter, I present and test a BBN model to diagnose factors limiting stream brown trout fisheries. I developed the BBN to help managers consolidate the vast body of literature available on brown trout and apply it in a practical context. Once limiting factors are identified, appropriate management actions can be taken to address them.
Underpinning the BBN is a substantial review of the salmonid literature that is presented in Appendix 1. When preparing the literature review, I could not follow the conventional literature review manuscript structure. Instead, the document structure was dictated by the requirements of the BBN modeling process. First, with the help of my supervisors and colleagues, I identified key variables that can potentially limit brown trout populations in streams (e.g. temperature, instream cover, nitrate concentrations). I then selectively scrutinised the literature to find numerical ranges and thresholds to define category breakpoints for the various potential limiting factors. The discretised value ranges for the variable categories were then used to define the limiting factor probabilities used within the BBN-LFA.

Serendipitously, the unusual document structure of Appendix 1 presents information in a useful format to inform limits-based freshwater management frameworks, which are currently being tested in New Zealand. The National Policy Statement for Freshwater Management (NPSFM) requires regional councils to set compulsory limits for water quality (e.g. nitrate concentrations and E.coli counts) and quantity (e.g. minimum flows) and then manage for them (NPSFM 2014). Within the NPSFM, there is provision for councils to set limits to protect voluntary instream values. In some catchments, trout fisheries may be considered an appropriate voluntary value to manage for. The literature review in Appendix 1 provides an up-to-date set of numerical values for key instream variables (or attributes) that ought to help managers set appropriate environmental limits to manage trout fisheries. More broadly, the Appendix 1 review demonstrates that managers will need to consider a wide range of environmental attributes in order to achieve community aspirations for productive stream fisheries—well beyond just managing for water quality and quantity, which currently are the focus of the NPSFM and freshwater management in New Zealand in general.

The BBN modeling approach is highly conducive to modeling freshwater fishery dynamics. High levels of complexity and uncertainty are inherent within
freshwater fisheries science, because data is sparse relative to other ecological
disciplines. Nonetheless tools are needed now to help managers make decisions.
And decisions should be based on the best information available—regardless of
the degree of uncertainty underpinning current theory. Indeed, in many areas of
brown trout ecology, developing the BBN-LFA required stretching scientific
knowledge and theory—but this is the reality of applied science. The BBN
modeling framework allows a flexible, pragmatic approach for incorporating
empirical data, ecological theory and expert opinion, whilst maintaining
transparency regarding uncertainty (Aguilera et al. 2011).

I tested the BBN-LFA on the Horokiri Stream. This data-rich catchment was the
focus of the landmark freshwater fisheries publication by Allen (1951): ‘A study of
a trout population’. Allen’s study meticulously documented trout population
dynamics in relation to food and habitat conditions over a decade. I used the BBN-
LFA to hind-cast predictions of the limiting factors in the Horokiri Stream, both in
its ‘reference’ state based on Allen’s data (from 1951) and in its current degraded
state. The current-state data were extracted from a more recent in-depth expert-
based assessment by leading fishery scientists using habitat and fish survey data
collected after the collapse of the fishery (Jellyman et al. 2000). The BBN-LFA
outputs were comparable with the expert-based assessment, demonstrating the
suitability of BBN modeling for undertaking a limiting factor analysis on stream
brown trout populations. More broadly, this chapter demonstrates that BBN
models can be used to predicted limiting factors for a range of aquatic values.

5.3. Chapter 3. Mechanically reshaping stream banks alters fish community composition (Holmes et al. 2019)

Bank reshaping (reshaping) is a common management action undertaken in
lowland agricultural streams to reduce excessive bank erosion. It involves using a
digger to scrape back the banks to a relatively shallow angle, usually to a 1 in 2
slope, so they are less prone to undercutting and mass failure. In 2014, a large-scale reshaping event was undertaken in Waituna Creek (Southland, New Zealand), along approximately 14 kilometres of both banks. The project was supported with funding from the New Zealand government’s ‘Fresh Start for Freshwater’ rehabilitation fund (http://www.mfe.govt.nz/fresh-water/clean-projects/waituna-lagoon). The goal of the reshaping project was to reduce the amount of fine sediment entering the ecologically sensitive Waituna Lagoon downstream, which has international conservation significance (Draft Waituna Plan 2015).

The rehabilitation project in Waituna Creek represented an opportunity to investigate the effects of reshaping on instream values. Among some river engineers and managers, it was perceived that the fish populations would be largely unaffected by reshaping because the low-flow wetted channel remains undisturbed during the process. To investigate this claim, and my own contrasting hypothesis that large-bodied fish would be negatively affected by the practice, I collected quantitative electric fishing and instream habitat data, two months before and annually for three years after the reshaping event. Data were collected using a Before-After-Control-Impact (repeated measures) study design as part of a collaborative effort with the Department of Conservation and other research partners.

In contrast to the stated aim of the project, the reshaping increased deposited fine sediment levels in Waituna Creek. One year after reshaping percent fine sediment cover increased by 31% at the impact sites, whereas, there was a 35% decrease at the control sites. I could not investigate the long-term effects that the reshaping project might have on deposited fine sediment levels. For example, it is plausible that over five to ten years fine sediment levels would decrease after the initial pulse, if the reshaping reduces ongoing bank erosion. Nevertheless, this result for the first few years after reshaping highlights the importance of monitoring rehabilitation projects because they can have unintentional, and in this case potentially counterproductive, consequences.
One year after the event, longfin eel biomass was reduced by 49% in the impact reaches, relative to a 160% increase in the control reaches. Sequential regression analysis suggests that hiding cover became a limiting factor for eels in the reshaped stream segments. In addition, the densities of common bullies showed a staggering increase in the impact reaches—by 4000 percent. The increased bully abundances might have been the result of reduced predation pressure from eels. However, eventually, three years after reshaping, fish community structure had largely returned to its pre-impact state.

This chapter shows that even in highly channelised streams, further bank reshaping can cause a substantial proportion of the eel population to migrate to other stream segments in the catchment—with flow-on effects to the stream food web. It is likely that habitat quality and the carrying capacity for eels was temporarily reduced by reshaping. In future, before consents for reshaping streams are granted, the potential fish community effects should be weighed against any perceived erosion control benefits. In common with the themes of Chapters 2, this research demonstrates that stream habitat attributes not directly related to water quality or quantity can limit fish carrying capacity in streams.

5.4. Chapter 6. Riparian management affects instream habitat condition in a dairy stream catchment (Holmes et al. 2016)

In this chapter, I use a space-for-time substitution study design to investigate instream responses to different riparian management practices within a spring-fed dairy catchment. The study occurred in the Waikakahi Stream (Canterbury, New Zealand), a decade after a catchment-scale riparian fencing and planting initiative was undertaken in response to severe degradation from dairy conversion and ongoing poor farming practices (Monaghan et al. 2009). As a result of this rehabilitation initiative, the Waikakahi Stream has a wide gradient of riparian
management area widths, from no riparian fencing to 25 m-wide stock exclusion areas on both banks. The spectrum of riparian management practices, combined with the homogeneous, spring-fed nature of the stream throughout its length (i.e. similar bank slopes, gradients and stable flows), provided a unique opportunity to investigate localised instream responses to different riparian management practices.

With the help of Central South Island Fish and Game staff and my Cawthron Institute colleagues, I collected representative samples of instream and riparian habitat data throughout the catchment using a catchment-scale habitat mapping protocol detailed in Holmes & Hayes (2011). Quantitative electric fishing data were also gathered from some of the instream survey sites. I used these data to investigate potential causal relationships between riparian management practices and instream ecological outcomes. This was done by comparing combinations of riparian and instream variables using sequential regressions at incrementally increasing spatial scales.

Most riparian and instream regressions showed no apparent correlation, indicating that the effects of the rehabilitation efforts may be subtle or non-existent for many aspects of instream ecology (e.g. trout and longfin eel densities or biomass), at least at the reach to segment scales. However, there was a significant negative correlation between deposited instream fine sediment cover and the area of stock exclusion fencing in upstream reaches. To make this result relevant for managers, I used the relationship to show that stream reaches (>300 m long) with 5 m-wide stock exclusion fences (both banks on average) were associated with instream fine sediment cover below 20 percent in reaches immediately downstream; a minimum rule of thumb considered appropriate to maintain healthy invertebrate and fish communities in streams (Clapcott et al. 2011). Therefore, in the Waikakahi Stream, and possibly other spring-fed streams, all fencing should be set back at least 5 m from the low-flow water’s edge—to reduce fine sediment downstream to levels that will improve stream health.
I acknowledge that the correlative nature of this study reduces its explanatory power. However, the result precisely fits with relevant theory and a common-sense analysis of the effects of fencing in small, low-gradient, spring-fed streams such as the Waikakahi. Since publishing this research in 2016, I have received correspondence from rural professionals questioning the generalisability of the 5-m minimum fence setback recommendation. As pointed out in the paper, this recommendation was proposed as a practical ‘rule-of-thumb’ for farm streams that are similar in nature to the study stream. Different streams will have different optimal setbacks. This is because variables such as bank slope, soil type, stream power and adjacent land use will combine to influence erosion and sediment transport dynamics in complex ways (Wood & Armitage 1999). Also, biota within different streams will require differing levels of protection, depending on their conservation value or susceptibility to fine sediment. Nevertheless, Chapter 4 places a pin on the spectrum of required fencing widths that appear to result in a measurable instream habitat benefit. The research also shows that broadscale stream habitat mapping is a useful (and practical) method for monitoring the effectiveness of riparian management actions for improving instream structural/physical habitat.

5.5. General thesis implications

Taken together, the first four chapters of my PhD thesis (including Appendix 1) highlight the importance of considering a broad range of attributes when assessing the ecological effects of habitat degradation or rehabilitation. In particular, physical/structural habitat can limit key instream values. This outcome is particularly relevant for current freshwater management frameworks within New Zealand, which largely ignore physical habitat attributes—although work is underway to include deposited fine sediment as an attribute in the NPSFM management framework (MFE 2018). Based on my research, I argue that management approaches that focus narrowly on water quality and quantity attributes will fail to address limiting factors for a host of biota and instream values—particularly those relating to higher trophic level fish such as eels and
trout. It is therefore vital that broader attributes that could potentially limit biotic responses, such as those listed in Appendix 1, are incorporated into freshwater management.

Another interesting concept to emerge from this PhD is the apparent stark disparity between the spatiotemporal scales of stream habitat degradation and rehabilitation. Stream degradation occurs at the landscape scale, whereas habitat rehabilitation is typically targeted at reach to segment scales (Hillman & Brierley 2005). Moreover, degrading an ecosystem occurs rapidly, assisted by the economic incentives to intensify land use. However, those who wish to reinstate ecological health in streams effectively find themselves working against the flow of contaminants (without an economic paddle). Yet, as I demonstrate in Chapter 4, if a catchment-scale approach is taken, with broad landowner buy-in and determined community leadership, some ecological success can be achieved within just a decade.

5.6. A stream management thought experiment

The following thought experiment is intended to highlight the disparity between the scales of stream habitat destruction and rehabilitation, as well as introduce the concepts of hysteresis and/or recovery lag as it has been applied in aquatic ecology. Here we define hysteresis as resistance of an ecosystem to recovery, following stress removal, because of internally mediated feedback loops. For example, stable phytoplankton dominates states in shallow lakes (Scheffer et al. 2001; Beisner et al. 2003). We define ‘recovery lag’ as a relatively slow and monotonic ecosystem responses to stress removal that lacks any internal forcing mechanisms. For example, macroinvertebrate population growth after a catastrophic flood (Townsend 1996).
My thought experiment involves considering the likely ecological outcomes that will result from two contrasting (hypothetical) stream habitat manipulation scenarios, as outlined below:

**Scenario One:**
Imagine a pristine low-gradient spring-fed stream in the middle of a vast conservation park—no human foot has ever been placed within its catchment. Now imagine that a team of engineers and ‘evil ecologists’ are helicoptered into the stream, mid-catchment. They use modern technology and ecological theory to precisely degrade a 100 m reach of stream. They clear the riparian vegetation, plant invasive macrophytes and riparian weeds, release invasive fish and invertebrates (but erect screens and electrical barriers to keep them within the reach). They replace the riffle-pool mesohabitat structure with a continuous homogeneous run and load fine sediment onto the stream bed. The stream banks are shaped into a uniform trapezoid channel. Half the flow is diverted around the reach and nutrients and contaminants are drip-fed into the remaining flow at the upstream end, so that contaminant concentrations are equivalent to those found in the most degraded urban/agricultural catchments. As water exits the downstream end of the reach the team run a water purifier that removes the added contaminants. Remember, all these interventions are limited in extent to a 100 m reach in the mainstem mid-catchment.

**Scenario Two:**
Now imagine the exact opposite scenario: An identical spring-fed stream, except in this case, the entire catchment has been intensively farmed for a century. Furthermore, land use practices have been ‘worst practice’. Heavy stock have had unrestricted access to the stream channel, dairy shed effluent is discharged directly to the headwaters, etc... In this degraded catchment, a team of engineers and ecologists are brought in to rehabilitate a 100 m reach in the mainstem mid-catchment. They fence out the stock, vacuum the fine sediment from the stream bed, reinstate a riffle-pool sequence and add vegetative habitat structures. They re-contour and replant the stream banks and pipe clean water into the reach to...
reinstate a natural flow regime. Somehow, the team manages to remove all land-use-derived water contaminants at the upstream end of the reach in real time, as well as adding those contaminants back to the downstream end. All these interventions faithfully represent the pre-development habitat conditions within this isolated 100 m reach.

**Compare and contrast**

Now I ask you to consider: how will the ecosystem and individual biota respond to these two contrasting scenarios? Would biota respond in the exactly opposite ways to mirror the exactly contrasting treatment scenarios? Below I argue that biota will not respond in opposing ways.

Under Scenario One, it is easy to imagine a rapid ecological response to the degradation treatment. Fish will migrate out of the degraded reach (Duvel et al. 1976; Jowett et al. 2009) and those sensitive to habitat changes and pollution will not come back (Frothingham et al. 2013). Macroinvertebrates that are intolerant of fine sediment will drift downstream (Matthaei et al. 2006) and benthic algal biomass will accrue rapidly, in a matter of weeks, in response to increased light and nutrients (DeWalle 2010). Overall, species diversity will drop as generalist pollution-tolerant taxa dominate (Niyogi et al. 2007), as will overall ecosystem health and functioning. Contrariwise, if pre-treatment habitat conditions were faithfully reinstated in the degraded reach, I suggest that there would be a rapid return to pre-treatment ecological conditions.

Under Scenario 2, by contrast, there will be a substantial temporal lag before ecological improvements accrue. Riparian plants will take time to establish and invertebrates and fish will take time to find and colonise the new ‘more suitable’ habitat that has been created. Moreover, species diversity may never increase because source populations may be severely depressed or non-existent because of stress occurring elsewhere in the catchment (Palmer et al. 2010).
This thought experiment is intended to highlight the fact that an ecosystem’s response to rehabilitation efforts will depend both on the scale and the history of environmental stress (Harding et al. 1998). Furthermore, it emphasises that scale is all important for restoring ecological processes (Bernhardt et al. 2011). In my Chapters 3 and 4, these scenarios were qualitatively approximated by the response, or lack of response, of fish populations to the respective ‘treatments’ that Waituna Creek and Waikakahi Stream received: subcatchment-scale degradation over six months in Waituna Creek vs. reach- and segment-scale rehabilitation over 10 years in Waikakahi Stream. In the latter, a decade on, I observed some improvement to instream habitat but no obvious spatially explicit response from trout or eels. In contrast, in Waituna Creek there was a relatively instant and strong response from the fish community to habitat removal.

The idea illustrated by this thought experiment is, in part, captured by the concept of hysteresis as it has been applied within the ecosystem rehabilitation literature (e.g. Scheffer et al. 2001). Below, I introduce a conceptual model that provides a framework for incorporating scale into hypotheses about river ecosystem recovery. The following section has been written as a standalone ‘mini-review’, with the intention of publishing it as a short communication.

### 5.7. Scaling ecological hysteresis

#### 5.7.1. The history of hysteresis

A mature family of theories now describe how freshwater ecosystems respond to stress. Concepts with wide explanatory power include: disturbance ecology (Resh et al. 1988; Townsend et al. 1998), ecosystem resistance and resilience (Allison 2004; Hildebrand & Utz 2015), cumulative effects (Schindler 2001), and more recently, multiple stress ecology (Ormerod et al. 2010). Yet the response of ecosystems to stress removal has received much less theoretical attention.
Hysteresis is a relatively recent concept in ecology, borrowed from physics, that describes the recovery lag of an ecosystem as being dependent on antecedent conditions (Scheffer et al. 2001; Beisner et al. 2003). Put another way, memory of an ecosystem’s degradation pathway is conserved within a system’s recovery trajectory through internal forcing or feedback loops. There are well documented examples of hysteresis occurring in shallow lakes (Dent et al. 2002) and coral reefs (deYoung 2008). However, evidence for hysteresis occurring in other aquatic ecosystems is sparse (Hildebrand & Utz 2015; Capon et al. 2015). Moreover, the external validity of the theory is hindered by a need for further concept refinement and integration with broader ecological theory (Ratajczak et al. 2018). For example:

1) how do the interactive effects of multiple stressors ‘disassociate’ as stressors are removed from an ecosystem and does this affect recovery trajectory?
2) how does the spatiotemporal scale of ecosystem stressor addition and removal affect hysteresis or resistance to recovery?

To encourage investigation into the latter question, in this short review paper I present a graphic framework that depicts how spatial scale might influence ecosystem recovery. This conceptual model was developed with river rehabilitation in mind; however, it also ought to apply to ecosystems more widely.

5.7.2. Ecosystem elasticity

In their landmark review paper, Lake et al. (2007) summarise four potential ecosystem recovery models (Figure 16). Below I describe these models and then argue they will gain further descriptive traction by incorporating the hierarchical river-scales concept (Frissell et al. 1986; Townsend 1996).
Figure 16. The four models of ecosystem recovery reproduced from Lake et al. 2007 (adapted from Sarr 2002). Model A shows the ‘rubber band’ response where the recovery of a system follows the same path as its degradation. B shows strong hysteresis where recovery lags substantially before eventually returning to the pre-existing state, C and D show the ‘humpty dumpty’ and ‘shifting target’ models where the recovery endpoint is different from the initial starting state and, in the case of D, the pathway and endpoint may be unpredictable.

Within a lotic context, Figure 16C (showing the ‘humpty dumpty’ response) could model the effects of forest clearcutting on spring-fed streams. Denuding a catchment of trees can lead to high fine sediment loading, with fines becoming embedded within the gravel matrix of the stream substratum (Wood & Armitage 1999). If the stream lacks power to mobilise embedded fines, this will lead to practically irreversible changes to the benthic invertebrate community—even if fine sediment supply rates are returned to pre-harvest levels (Gayraud et al. 2002; Lake et al. 2007). Figure 16D could describe a scenario where an environmental stressor has caused a permanent (localised) keystone species extinction (Bond 1994). Removing the stressors that caused the local extinction will not restore this
component of community structure—or the stabilising function that the organism may have had on the food-web (Jordan 2009; Burchsted et al. 2010).

The ‘rubber band’ and ‘hysteresis’ response shapes (Figure 16A, B) depict more generalisable recovery models, where the initial ecosystem state is attainable, at least eventually. The hysteresis ecosystem response is perhaps best demonstrated by the internal forcing mechanisms that can maintain stable phytoplankton-dominated states in shallow eutrophic lakes (Scheffer et al. 2001). For example, high turbidity caused by a phytoplankton bloom can shade macrophytes, causing senescence, which allows for the resuspension of nutrients held in lakebed sediments to fuel further phytoplankton growth. In turn, this strengthens the stability of the ‘turbid state’.

In comparison with lakes, river ecosystems are thought to be more dynamic and only weakly (if at all) controlled by internal or endogenous processes. This is because they are dominated by external physical forces such as floods, which can regularly ‘re-set’ a system (Resh et al. 1988; Townsend et al. 1989). Nevertheless, Dent et al. (2002) provide a convincing argument for potentially important, endogenously maintained stable states occurring in rivers. For example, nutrients supplied to headwater streams by rotting salmon carcasses from large spawning runs may create more productive juvenile salmonid rearing habitat. A poor salmon run in one year could leave the system vulnerable to a negative feedback loop, because the lack of nutrient subsidy from spent salmon could lead to poor recruitment and spawning run strength in subsequent years (Stockner 2003; Compton et al. 2006). Dent et al. (2002) also point out, that the potential for hysteresis may vary with ecosystem stability, irrespective of whether the ecosystem in question is a river or lake. For example, small shallow lakes can be unstable environments if they are prone to periodic drying and conversely, spring-fed streams can be very stable systems (Townsend 1989).
5.7.3. Ecosystem recovery across river scales

The models of ecosystem recovery discussed above do not explicitly incorporate spatial scale. Groffman et al. (2006) suggests the issue of spatial scale sits implicitly and “uncomfortably” behind most theories attempting to describe how ecosystems respond to stress removal. Yet in practice, numerous authors suggest that many rehabilitation projects produce underwhelming results precisely because they are applied at relatively modest scales (e.g. Beechie et al. 2008; Hillman & Brierley 2005). The field of dreams hypothesis: ‘if you build it, they will come’ (Palmer et al. 1997), has seldom been realised because projects often fail to account for catchment-scale stressors (Palmer et al. 2010).

Figure 17 graphically illustrates the hypothesis that increasing the spatial scale of degradation will increase recovery resistance in rivers during stress removal. The figure is a combination of the concepts described by Lake el al. (2007) and the hierarchical river scale concepts described in Frissell et al. (1986), Townsend (1996), Maddock (1999), and Leemput et al. (2015). Figure 17 should be considered alongside the conceptual model shown in Figure 18 (Maddock 1999). This model shows how the time scale of an ecosystem response might be related to the spatial scale of catchment pressure.
Figure 17. The scale-dependent ecosystem recovery model. A new conceptual model showing ecosystem degradation and recovery trajectories in response to stress as a function of river scale—from a stream patch to an entire catchment. The open end of the envelope (left) shows intense resistance and recovery and/or hysteresis as a result of catchment-scale degradation and stress removal. At small spatial scales (right), an ecosystem may show little or no resistance to stress addition or removal (i.e. a more ‘rubber band’ type response occurring at the patch scale). The x and y scales are invariant across the z axis.
Figure 18. A conceptual model adapted from Maddock (1999) that proposes how a river ecosystem’s sensitivity to change is related to space (left) and time (right).

The shape of degradation and recovery trajectories, and the degree to which scale influences resistance to recovery, will be highly dependent on the type of ecosystem stress. In Figure 17, I chose a simple exponential function to represent some initial ecosystem resistance. However, it must be noted that ecosystem responses to some stressors, such as fine sediment, can show little or no resistance, at least along the degradation pathway (Matthaei et al. 2006). Using the example of deposited fine sediment again, if fines are added to a patch or reach, and then removed, the benthic macroinvertebrate recovery response will be rapid and will closely follow the degradation pathway (Culp et al. 1986; Melo et al. 2003; Matthaei et al. 2006; Ramezani et al. 2014). By contrast, if high levels of deposited fine sediment occur at segment or catchment scales (Buendia et al. 2013), then Figure 17 and 18 predict sluggish and more intense resistance and/or resilience to ecosystem recovery following its removal (Burnhardt & Palmer 2011). In the latter instance, during recovery in rehabilitated areas, potential source populations of sediment-sensitive macroinvertebrates would be absent from a high proportion of neighbouring habitat. Furthermore, some degree of resistance can be expected as colonists of restored habitat undergo exponential population
growth. Therefore, macroinvertebrate community recovery could be expected to be hindered in proportion to the amount of impacted habitat.

While this concept may seem intuitive, when applied to regional-scale freshwater management somewhat counterintuitive recommendations emerge, as explained below.

5.7.4. Building ecosystem history into future management

The spatial-scale dependent ecosystem recovery model explicitly places ‘low hanging fruit’ for rehabilitation within near-pristine catchments—rather than within catchments that already have widespread degradation (Johnson et al. 2016). This concept is an extension of the intuitive notion that prevention is more resource-efficient than cure. Conversely, within regional management, it is common to assign relatively permissive pollution loads to degraded catchments in favour of protecting less disturbed ecosystems (Gillon et al. 2016). However, if the severity of ecological degradation is assessed by the amount of resources required to reverse it, then increasing the extent of degradation in already degraded catchments is potentially misguided. This is because more extensively stressing degraded systems could disproportionately reduce the ‘reversibility’ of any negative ecosystem response, especially if the system is pushed past a tipping point (Amoros et al. 1987).

My conceptual model is put forward to help frame experimental inquiry into ecosystem hysteresis and/or resistance to recovery in rivers. In particular, studies are needed to define how systems respond to rehabilitation efforts within riverscapes that vary in degradation extent. There is a pressing need for predictive models that can guide the scale at which rehabilitation tools should be applied in order to be effective. Such research is necessary to enable the sparse ecosystem improvement resources to be more effectively allocated within and between catchments.
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7. Appendices

Appendix 1. A literature review to support a limiting factor analysis of stream brown trout populations. Here we present a literature review to underpin a Bayesian Belief Network-based (BBN) limiting factor analysis (LFA) for stream brown trout fisheries. The BBN model is described in detail in Chapter 2. The primary purpose of this literature review is to: 1) determine key environmental variables that can potentially limit stream populations of brown trout and 2) define numerical ranges and thresholds for those environmental variables that would support different trout population levels—from non-existent to thriving.
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1. A literature review to support a limiting factor analysis of stream brown trout populations

1.1. Introduction

The information in this review is used to inform ‘break points’ within the BBN models’ environmental data input nodes (parent nodes). We also use the information to define the conditional probability tables (conditional probabilities) that link the BBN parent nodes with the ‘child nodes’ (Marcot et al. 2006). The probabilities define when a trout population is likely to be limited if an environmental variable sits within certain range or ‘bin’. Figure 1 shows a simplified example of the BBN model structure. A detailed background, methodology and an example application of the BBN to the Horokiri Stream (Wellington, New Zealand) is presented in Chapter 2 and in Holmes et al. (2018).
Figure 1. The basic structure of the Bayesian Belief Network limiting factor analysis model (left) and example (right). In the top row are the root parent nodes, which contain some environmental parameter information. In the next row are causally linked nodes which are conditional on the parent nodes. Below these are the horizontally listed limiting factor probability nodes. In the bottom row are the diagnostic test nodes, which contain fishery population-health metrics. An example of a simplified sub-net for ‘Food Limitation’ is shown on the right.

The literature review is structured as follows. For each potential limiting factor variable within the BBN we:

1) briefly introduce the variable and explain why it was included (or excluded) from the BBN
2) list the key environmental-correlate nodes (parent nodes) that comprise the relevant sub-network branch of the BBN
3) suggest the form of data that should be used to populate the relevant parent nodes and how to collect it
4) explain how we determined environmental variable category-breakpoints for the relevant parent nodes
5) detail how the values in the BBN conditional probability tables were determined for each limiting-factor child node.

As well as supporting the BBN, this literature review may also be useful for a variety of salmonid management applications. For example, it would allow determining environmental attribute limits that provide for the maintenance of brown trout fisheries in streams.

Our suggestions for the data requirements to populate the BBN parent nodes are not intended to be fully prescriptive. Pre-existing data, collected using different protocols, may be useful if they are reinterpreted with respect to our input data suggestions. However, identical data collection methodologies should be used if comparing limiting-factor probabilities over time or between sites.

Based on our review, the key limiting factors that we chose to include in the BBN are shown in Table 1.
Table 1. Key limiting factors for brown trout (*Salmo trutta*) in New Zealand wadeable lowland streams and their binary logic statement codes used in the BBN. Factors are ranked in a hypothetical order according to their likelihood of limiting trout production in a ‘typical’ New Zealand stream. Rankings are based on our interpretation of the literature.

<table>
<thead>
<tr>
<th>Key factors that limit stream brown trout fisheries</th>
<th>Binary logic statement codes used within the BBN</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Temperature</td>
<td>Too hot</td>
</tr>
<tr>
<td>2. Flow (flood flows, low flows)</td>
<td>Flood-limited, Low-flow-limited</td>
</tr>
<tr>
<td>3. Recruitment</td>
<td>Recruitment-limited</td>
</tr>
<tr>
<td>4. Food</td>
<td>Food-limited</td>
</tr>
<tr>
<td>5. Instream fish cover</td>
<td>Cover-limited</td>
</tr>
<tr>
<td>6. Water quality</td>
<td>Water-quality-limited</td>
</tr>
<tr>
<td>7. Angler-usage</td>
<td>Usage-limited</td>
</tr>
</tbody>
</table>

The information collected as part of this review is summarised in Table 2 where we give the rationale for including or excluding each potential limiting factor, suggest data requirements to populate BBN parent nodes and state the knowledge source(s) used to define breakpoints for parent node categories. Once the data requirements detailed in this Appendix are met, data can be entered into the Netica BBN model (available from the author on request). Guidance on interpretation of model results, by way of an example application, is given in Chapter 2.
Table 2. Summary table for the decisions inherent within our Bayesian Belief Network (BBN) for undertaking a limiting factor analysis (LFA) on a stream brown trout population. For each limiting factor child node, the immediate environmental variable (parent) nodes are given along with a description of the parent node data requirements and their respective weightings used to calculate conditional probabilities. Also provided is our subjective assessment of how robust / adequate the available scientific knowledge base is that underpins the related decisions made for each BBN sub-net (very low, low, moderate or high).

<table>
<thead>
<tr>
<th>Limiting factor</th>
<th>Immediate parent nodes</th>
<th>Parent node discretisation thresholds</th>
<th>Knowledge source</th>
<th>Variable weight</th>
<th>Degree of confidence in knowledge base</th>
</tr>
</thead>
<tbody>
<tr>
<td>Recruitment limited</td>
<td>Rolling average Number of August – November floods that exceed 10*median flow (FRE10) every three years</td>
<td>&lt;1, 1 – 3, &gt;3</td>
<td>Literature, logic</td>
<td>Weight 1</td>
<td>High</td>
</tr>
<tr>
<td>Catchment spawning and juvenile rearing habitat</td>
<td>Good, Ok, Poor</td>
<td>Expert opinion</td>
<td>Weight 0.5</td>
<td>Very low</td>
<td></td>
</tr>
<tr>
<td>Winter maximum temperature</td>
<td>≤ 10°C, &gt;10 °C</td>
<td>Literature</td>
<td>Weight 0.5</td>
<td>High</td>
<td></td>
</tr>
<tr>
<td>Large lake in the system</td>
<td>True or False</td>
<td>Literature, expert opinion, logic</td>
<td>Weight 0.5</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Food limited</td>
<td>Forage-fish supply Present or absent</td>
<td>Literature, national spatial database, continuous variable</td>
<td>Weight 0.5</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Macroinvertebrate density &lt;750 / m², 750 – 3000 / m², &gt;3000 /m²</td>
<td>Literature, expert opinion, quantile</td>
<td>Weight 0.5</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Metric</td>
<td>Category</td>
<td>Method</td>
<td>Weight</td>
<td>Confidence</td>
<td></td>
</tr>
<tr>
<td>--------------------------------</td>
<td>--------------------------------------------------------------------------</td>
<td>-----------------------------------------</td>
<td>--------</td>
<td>------------</td>
<td></td>
</tr>
<tr>
<td>Drift-feeding opportunity metric</td>
<td>High or Low (BBN sub-network)</td>
<td>Literature, expert opinion</td>
<td>0.5</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Temperature</td>
<td>Too hot or not too hot (BBN sub-net)</td>
<td>BBN network sub-branch (see below)</td>
<td>1</td>
<td>High</td>
<td></td>
</tr>
<tr>
<td>Recruitment</td>
<td>Recruitment limited Yes / No (BBN sub-net)</td>
<td>BBN network sub-branch (see below)</td>
<td>1</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Low-flow limited</td>
<td>Coefficient of flow variation &lt;1, 1 – 3, &gt;3</td>
<td>Technical report</td>
<td>1</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Percentage of reach ≥ 1m deep at base flow</td>
<td>&lt;10%, 10 – 50%, &gt;50%</td>
<td>Literature, visual assessment of data distribution for break points</td>
<td>1</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Flood-flow limited</td>
<td>Average number of FRE10 floods per year ≤ 3.5, 3.6 – 4.7, 4.8-7.5, &gt;7.5</td>
<td>Literature, expert opinion, unpublished data</td>
<td>1</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Segment gradient (%)</td>
<td>≤ 1.4, 1.5 – 4, greater than 4</td>
<td>Literature, expert opinion,</td>
<td>1</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Fish-cover limited</td>
<td>True, false</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fish-cover limited</td>
<td>Percentage of reach ≥ 1m deep at base flow &lt;10%, 10 – 50%, &gt;50%</td>
<td>Literature, expert opinion, visual assessment of data distribution for break points</td>
<td>1</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>Undercut banks</td>
<td>0% of bank, &gt;0 -50%, &gt;50%</td>
<td>Literature, expert opinion, visual assessment of data distribution for break points</td>
<td>0.75</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Large wood</td>
<td>&lt;5% of wetted area, ≥ 5%</td>
<td>Expert opinion, logic</td>
<td>0.75</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Overhanging vegetation</td>
<td>0% of bank, &gt;0 -50%, &gt;50%</td>
<td>Literature, expert opinion, visual assessment of data distribution for break points</td>
<td>0.5</td>
<td>Low</td>
<td></td>
</tr>
</tbody>
</table>
The distribution of trout and the productivity of stream fisheries is determined by water temperature regimes, both within and between catchments (Jowett 1990; Hayes et al. 2000; Armstrong et al. 2003; Elliott & Elliott 2010). For example, in lowland New Zealand, the northern extent of brown trout occurrence is limited by high summer water temperatures (Scott & Poynter 1991). Accordingly, consideration of a stream’s temperature regime, in particular the upper temperature ranges that fish may experience, is a critical component of any LFA for a trout fishery.

To maintain simplicity within the BBN, we chose not to include a limiting-factor node to account for temperatures below optima for brown trout. The productivity of practically all brown trout streams is limited by low water temperatures during winter and spring low temperatures (e.g. <10°C), which maintain trout metabolic rates below optima for processing food. Lower-range temperatures are largely

<table>
<thead>
<tr>
<th>Water quality limited</th>
<th>Dissolved oxygen</th>
<th>0 – 4 mg/L, 4 – 6 mg/L, &gt;6 mg/L</th>
<th>Literature</th>
<th>Weight 1</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>NO₃</td>
<td>&gt;8 mg/L, 4 – 8 mg/L, 0 – 4 mg/L</td>
<td>Literature</td>
<td>Weight 1</td>
<td>Moderate</td>
<td></td>
</tr>
<tr>
<td>pH</td>
<td>5.5 – 8.5 pH, outside this range</td>
<td>Literature</td>
<td>Weight 1</td>
<td>High</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Underutilised fishery – social limitation</th>
<th>Algae % cover</th>
<th>&lt;35% composite mat (&gt;3mm) and filamentous cover, ≥ 35%</th>
<th>Technical report</th>
<th>Weight 0.66</th>
<th>Moderate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Macrophyte % cover</td>
<td>&lt;10% cover, 10 – 20%, &gt;20%</td>
<td>Technical report</td>
<td>Weight 0.33</td>
<td>Low</td>
<td></td>
</tr>
<tr>
<td>Contact recreation status</td>
<td>Swimmable, secondary contact, no contact</td>
<td>NZ Government, Ministry for Environment derived values</td>
<td>Weight 1</td>
<td>Moderate</td>
<td></td>
</tr>
</tbody>
</table>

### 1.2. Temperature (‘Too hot’ limiting-factor child node)

The distribution of trout and the productivity of stream fisheries is determined by water temperature regimes, both within and between catchments (Jowett 1990; Hayes et al. 2000; Armstrong et al. 2003; Elliott & Elliott 2010). For example, in lowland New Zealand, the northern extent of brown trout occurrence is limited by high summer water temperatures (Scott & Poynter 1991). Accordingly, consideration of a stream’s temperature regime, in particular the upper temperature ranges that fish may experience, is a critical component of any LFA for a trout fishery.
determined by latitude, altitude, topography, ground-surface water dynamics, stream channel aspect and riparian vegetation (Poole & Berman 2001). Other than riparian vegetation removal, which generally results in undesirable ecosystem outcomes (Slevers et al. 2017), there are no practical management levers to address these variables.

1.2.1. Parent node(s) and input data

The too-hot limiting-factor node has one immediate parent node that requires long-term stream temperature data to populate. Since the advent of inexpensive, continuous-temperature loggers, temperature is now a simple variable to measure within a river segment. We suggest using long-term stream-segment temperature data, covering for example, at least five years of continuous records synthesised from 15 minute interval data. Various thermal regime summary metrics can be calculated from these data.

Davies-Colley et al. (2013) reviewed the ecosystem effects of temperature in New Zealand streams as part of the National Objectives Framework (NOF) initiative (MFE 2014). The NOF initiative is tasked with setting management objectives based on environmental attribute limits that maintain the ‘life supporting capacity’ of freshwater ecosystems. Davies-Colley et al. proposed that the Cox-Rutherford index calculated by: \[
\left(\frac{\text{Annual max temperature} + \text{mean temperature of the five hottest days of the year}}{2}\right)
\] was the most appropriate and ecologically relevant temperature summary statistic to define a river temperature ‘attribute’. It indicates sustained annual upper temperatures over a period of days. Consequently, this index is suited to indicate temperature conditions that may induce nonlethal effects on aquatic biota (e.g. modified behaviour). We chose the Cox-Rutherford index to inform our too-hot limiting-factor node based on Davies-Colley et al.’s rationale, and to align our model with data sets that are being generated to inform the NOF.

1.2.2. Data input categories

Brown trout can metabolise food at temperatures between 3.8 and 21.7°C (Elliott 1994; Elliott et al. 1995). However, food availability (both quantity and quality)
dictate the optimum temperatures for growth. Trout experimentally fed on maximum invertebrate rations exhibit optimum growth rate at around 14°C. When feeding on a more energy-dense fish diet (such as smelt), the optimum temperature for growth is around 17°C. At temperatures above 17°C growth potential declines rapidly, and temperatures above 23°C cause chronic stress. Acute mortality occurs when temperature exceeds 27°C (Elliott 1994; Elliott & Hurley 1999). Based on these temperature thresholds, Cox-Rutherford index values below about 19°C would present a low risk of behavioural disturbance in trout for substantial periods of the year. Maximum Cox-Rutherford index values in the range 19–24°C would be tolerable but metabolism would be impaired and feeding behaviour would be disturbed (Elliott 1994).

The temperature attribute ‘bands’ recommended by Davies-Colley et al. (2013) for the NOF were largely set with reference to rainbow trout thermal tolerance values. These values are equivalent to the thresholds that cause behavioural disturbance or mortality for brown trout. Therefore, we chose the draft NOF stream temperature bands (>19, 19–24, 24–27, >27°C) to define the category breakpoints for the single too-hot limiting-factor parent node in our BBN. In addition, proposed breakpoints are also supported by initial analysis of data from the Cumulative Effects Programme (CEP) survey streams (see Chapter 2 methodology for an explanation of these data), where juvenile trout (<200 mm) density (and biomass) showed a strong correlation with the Cox-Rutherford Index across relevant stream temperature bands.

1.3. Flow

Within rivers, flow is considered the ‘master variable’ because it is inextricably linked to all instream variables and processes (Poff et al. 1997). Fundamentally, a river’s flow regime, in combination with topography, defines the amount of space for fish (Jowett & Richardson 1989; Jowett 1997). Flow also dictates the rate of energy flux (organic sediment, debris and invertebrates) through a reach (Fausch
1984; Hayes et al. 2016; Naman et al. 2016). Simplistically, there are three key aspects of a flow regime that are relevant to trout habitat and invertebrate food supply. These include low flows for space, flushing flows to control algae and fine sediment and flood flows to define channel morphology (Jowett & Richardson 1989; Jowett et al. 2008; Crow et al. 2013; Fuller et al. 2013). Of particular relevance to our LFA are flood flows and extreme low flows which can be considered to represent disturbance events for a trout population.

1.3.1. Flood-limited node

Large floods displace and kill trout. Juvenile life history stages are particularly vulnerable (Harvey 1987; Strange et al. 1993; Hayes 1995; Holmes et al. 2013; George et al. 2015), although substantial floods can also severely impact adult trout populations (Allen 1951; Jowett & Richardson 1989; Young et al. 2001). The productivity of many fisheries in New Zealand rivers is likely to be limited by frequent flooding (Jowett 1990; Jowett & Duncan 1990). Nevertheless, it is hard to disentangle the direct effects of floods on trout populations from other potential limiting factors that are also linked to flooding. For instance, food resources may be poor in flood-prone rivers because invertebrate densities are low as a result of regular disturbance of stream substratum (Clausen & Biggs 1997; Matthaei et al. 1999; Melo et al. 2003; Effenberger et al. 2006; Olsen et al. 2013). Moreover, some populations of juvenile trout appear remarkably resilient in their response to floods. For example, a juvenile trout population in the Rainy River (Nelson, New Zealand) achieved typical densities eight months after a channel-defining flood with a 1-in-50 year return period that severely reduced autumn abundance of young-of-the-year. This was a result of density-dependent loss to the yearling stage (Hayes et al. 2010).

Stream slope and the structural complexity of habitat interact with stream discharge to determine the power of floods to displace or kill fish. High-gradient streams concentrate flows and have intense flushing power. Conversely, in low-gradient streams, high flows dissipate over a wider channel cross section, reducing near-bed velocity and the potential for disturbance to trout populations and their
food base (Jowett 1990; Poff et al. 1997). Structurally complex habitats (e.g., bedrock-lined pools) provide more diverse water velocities during floods and are refuges from high currents, flood debris and entrained sediment. Velocity refugia probably contribute to the resistance of trout populations to floods if extensively available in a stream (Penaluna et al. 2015).

1.3.2. Parent node(s) and input data

The ‘flood-limited’ child node has three parent input nodes: (1) flood-frequency, (2) segment-slope and (3) fish-cover (Table 2).

We used the mean annual frequency of flows that exceed 10 times the median flow (FRE10) as the index to represent the flood-frequency node. The FRE10 is commonly used to indicate flow stability. Clausen & Biggs (1997) showed that in most rivers a FRE10 event was enough to disturb a substantial portion of the substrate. Therefore, flows above this magnitude also have to potential to displace or kill trout, especially juveniles. Segment slope can be surveyed directly or estimated from the proximate segment(s) of interest from spatial mapping data bases; for instance, in New Zealand the River Environment Classification (REC) data base has modeled slope data for all stream segments (Snelder et al. 2004).

To approximate the degree of structural complexity within in a stream segment, we linked the fish-cover-limited node to the flood-limited node. The fish-cover-limited node is described in Section 1.9. It rates the probability-of-occurrence of structural fish security cover (e.g. woody debris), which is equivalent to flood refugia within a stream segment.

1.3.3. Data input categories

We determined category breakpoints for the flood-frequency node by quantile classification of the average frequency of FRE10 events within the CEP survey streams (see Chapter 2 methodology for an explanation of these data). For the segment-slope node, we based the category breakpoints on inflection points in
the empirical model by Kozel et al. (1989). This model showed that trout biomass is negatively correlated with stream segment slope. The fish-cover-limited node, which we assume substitutes for flood-refuge habitat, was linked to the flood-limited node as a continuous variable (i.e. with no category breakpoints). We weighted the parent nodes flood-frequency and segment-slope twice that of the fish-cover-limited node before calculating the conditional probabilities for the flood-limited node. We did this because we consider that the former two variables have an overriding influence on the ability of a population to resist floods, relative to reach-scale flood-refuge features.

1.4. Flow variability (low-flow-limited)

Low flows, either naturally occurring or exacerbated by water abstraction, affect fish populations first through indirect mechanisms before directly limiting abundance through restricting space (Armstrong et al. 2003; Harvey et al. 2006). The potential consequences of a low-flow event include reduced food availability (resulting from diminished quantity and quality of benthic invertebrate habitat and invertebrate drift), or poor water quality because of increased temperatures and reduced dissolved oxygen concentrations (Armstrong et al. 2003). Stressful conditions for fish that occur during annually reoccurring low-flow events are accounted for in other parts of the BBN. This is because we require that input data for the fish cover- (including depths), food- and water quality-limitation sub-network nodes are collected during flows near the 7-day mean annual low flow (7d-MALF). However, extreme low flows (i.e. droughts) can be considered disturbance events in their own right (Crow et al. 2013). Droughts can depress a trout population so that in subsequent years, with higher base-flows, available resources will be underutilised (Armstrong et al. 2003; Harvey et al. 2006; Leprieur et al. 2006; Strange et al. 1993).

Channel morphology interacts with low flow to determine the degree of stress fish populations experience. Adverse effects of drought will be ameliorated when stream morphology is such that deep refuge habitats are available at low flow—provided that water quality is adequate (e.g. Jowett & Richardson 1989).
1.4.1. Parent node(s) and input data

The low-flow-limited child node has two parent nodes: (1) the coefficient of flow variation (flow-CV) and (2) the percentage of water deeper than 1 m at base flow (% depth >1 m).

We did not choose an absolute measure of low flow. We needed a transferable flow metric that indicates if a fish population is likely to be prevented from fully utilising habitat at typical flows because of a space squeeze occurring during extreme low flows. Habitat availability indices estimated with hydraulic-habitat models (Armour & Taylor 1991) would be informative, but we rejected this approach because it is too resource-demanding to be included in our LFA framework. Instead we opted for a measure of overall flow variability.

There are many potential indicators of flow variability for streams, all of which are highly correlated. We chose the coefficient of flow variation because it can be easily calculated from long-term flow statistics (e.g. >10-year data sets).

We included a requirement to estimate the percentage of water depths >1 m at flows close to the 7d-MALF. Water depths over a metre can be considered suitable refuge habitat for large brown trout (DeVore & White 1978; Raleigh et al. 1986; Bjornn & Reiser 1991). Depths should be estimated within three randomly selected 100 m reaches within a target segment. Rapid-field-survey protocols for obtaining stream depth estimates can be found in Holmes & Hayes (2011), and an example application is provided in Holmes et al. (2016).

1.4.2. Data input categories

In general, rivers with a flow CV of <1 can be considered to have a relatively stable flow regime, with a high proportion of base flow, whereas those with a flow CV of >3 have highly variable flow regimes (MFE 1998; Jowett & Duncan 1990). We used
these CV breakpoints to define the three categories (<1, 1–3 and >3) for the flow-CV parent node.

We determined three categories (<10%, 10–50%, >50%) to define the %-depth greater than 1 m node by visually identifying breakpoints within the frequency distribution of the depth data from the CEP streams. The lower depth category gives a crude estimation of the lower range of the percentage of ‘deep water’ available at MALF in New Zealand lowland wadeable streams that support adult trout populations and fisheries (represented by the CEP data). Both the flow-CV and % depth greater than 1 m nodes were weighted equally to calculate the conditional probabilities for the low-flow-limited child node.

1.5. Juvenile recruitment

Any LFA for a fishery must consider the supply of recruits. Within our BBN, recruitment is defined as the supply of juvenile trout less than three years of age. This incorporates the spawning, fry and juvenile rearing life history stages (briefly explained below).

Brown trout construct redds in coarse gravels and small cobbles in which to incubate their eggs. These are usually located in the tails of pools and the edges of runs where the bed slopes upward in the direction of the current. In these areas, downwelling forces oxygen-rich water through the gravels (Louhi et al. 2008). Fry require relatively slow water usually along the bank edges. Stream edge vegetation helps creates pockets of slow water and can produce good salmonid fry rearing habitat (Heggenes et al. 1999; Armstrong et al. 2003; Bardonnet et al. 2006). Juvenile trout tend to occupy cobble-boulder riffles and runs with a diverse range of velocity microhabitats (Armstrong et al. 2003). Commonly, spawning and juvenile rearing habitat may be located tens of kilometres upstream of lowland-stream segments in relatively low-gradient headwaters. Headwaters are less impacted by land use pressures, and critically, embryo incubation temperatures are more likely to remain below 10°C (Jowett 1990).
Recruitment of trout populations to a given stream segment depends on the amount of spawning and rearing habitat relative to the amount of adult habitat within the catchment. In addition, recruitment can be highly variable from year to year, even in streams where there is ample spawning and rearing habitat. Floods, particularly during the critical fry emergence period in early spring, are a common cause of poor recruitment years in rivers (Ottaway & Forrest 1983; Hayes 1995; Jensen & Johnsen 1999). If successive annual floods occur during fry emergence, this could create a negative feedback loop where recruitment is further reduced because of low numbers of returning adult spawners. On the other hand, the presence of a lake or large wetland in a catchment will ameliorate the negative effects of floods on recruitment (Jowett 1990). These provide productive refuge habitats for displaced trout and dampen the power of floods downstream.

1.5.1. Parent node(s) and input data

The recruitment-limited node has four parent nodes: (1) a subjective assessment of the adequacy of spawning and rearing habitat (rearing-habitat node), (2) floods during the fry emergence period (spring-floods), (3) maximum winter temperature in spawning areas (max-winter-temp) and (4) the presence (or absence) of a lake below rearing areas.

Knowledge of the adequacy of recruitment within a catchment will vary widely between fishery management staff or fishery researchers and catchments depending on staff experience and the availability of fish population data. To ensure the qualitative assessment component, required to populate the rearing-habitat node, is as robust as possible as many independent assessments as feasible should be undertaken, with the results assessed for variation (see Marcot et al. 2006 and McDonald et al. 2015 for guidelines for incorporating expert assessments into BBNs). If there is inadequate knowledge available to populate this node, then when defining the BBN node settings, it should be held neutral and...
set to ‘OK’ to remove its influence on subsequent probability calculations within the BBN.

We chose to incorporate a parent node that represents the average occurrence of successive flood events in the spawning areas through August and September (New Zealand late winter–early spring). This node (spring-floods) accounts for floods during the critical fry emergence period. To populate the spring-floods node, we used the rolling-average number of spring FRE10 events within each three-year period of a flow record (ideally at least nine years). To account for the potential for lethal high temperatures during embryo incubation (i.e. in excess of 10°C), we included the max-winter-temp node. Finally, we included a parent node for the presence or absence of a substantial lake or wetland (e.g. >10 hectares) within the catchment—to account for the positive effect that lakes or wetlands can have on trout survival (Table 2).

1.5.2. Data input categories

In any subjective assessment of spawning and rearing potential in a catchment, there is a large degree of inherent uncertainty. Accordingly, we used just three broad categories (good, OK and poor) to define the rearing-habitat node. The three categories that define the spring-floods node were based on the reasoning that if spring floods occurred annually throughout a cohort’s three-year life cycle this would result in very poor recruitment. This is because trout typically mature within 3 years in New Zealand streams (McDowall 1990). If spring floods occurred twice or ≤ once every three years, this would provide for adequate or good recruitment, respectively (all other factors held equal). We used two categories, <10°C or ≥ 10°C during June, July or August to define the breakpoint for the max-winter-temp node. To define the node that represents the presence (or absence) of a lake or wetland below rearing areas in the catchment we used binary categories for the presence or absence of a lake or wetland in the system.

We weighted the parent nodes spring-floods, rearing-habitat, max-winter temp and the ‘presence of a lake/wetland’ at 1, 0.5, 0.5 and 0.5, respectively, before
calculating conditional probabilities. This reflects our opinion, backed by the literature cited above, that the occurrence of spring floods will be the dominant factor in determining recruitment potential in most catchments.

1.6. Food

Brown trout display varied feeding strategies to match their environment. Broadly, feeding strategies can be categorised into drift feeding (on aquatic and terrestrial invertebrates), benthic browsing and piscivory (Budy et al. 2013; Piccolo et al. 2014; Milardi et al. 2016). To some extent, food availability limits the production of trout in all wild fisheries. Taken to the extreme, this point is illustrated by the high biomass and large size of salmonids attainable in commercial trout farms that use energy-dense food (Piccolo et al. 2014). As a practical definition for our LFA, we consider food limitation as a food level that leads to growth rates below what can be achieved under maximum invertebrate rations, given typical temperature regimes in New Zealand rivers. Potential growth rate can be estimated with trout growth models (e.g. Elliott et al. 1995; Elliott & Hurley 1999, 2000; and also Hayes et al. 2000; Hayes 2013, which include Elliott’s equations).

If prey fish are plentiful and easy to obtain, they will be the preferred diet of trout (McCarter 1986; Keeley & Grant 2001). Trout feeding on fish diet can grow three times faster than on an invertebrate diet (Elliott & Hurley 2000) and therefore can obtain greater size and reproductive potential. Forage fish (including juvenile trout) are available as potential prey for larger trout in most rivers. However, in small wadeable streams, prey fish are assumed to be less important as food in comparison with stream invertebrates because of their abundance and ease of capture (e.g. Bachman 1984; Kelly-Quinn & Bracken 1990; Cunjak & Power 1987; McLennan & MacMillan 1984; Glova & Sagar 1991). In New Zealand streams, opportunities for trout to exploit fish prey will increase closer to the coast because of the prevalence of diadromy in the native fish fauna. Diadromous, pelagic fish species, such as īnanga (Galaxias maculatus) and common smelt (Retropinnidae),
can be especially abundant, at least seasonally (McDowall 1990). In stream segments that are near to the coast, piscivory may compensate for poor invertebrate food resources.

Drift feeding on invertebrates is a common feeding strategy for stream trout (Hayes & Jowett 1994; Fausch et al. 1997). Drift-feeding trout conserve energy by holding station in ‘slow’ water whilst intercepting drifting invertebrates from adjacent fast water (Fausch 1984). With the potential exception of some heavily forested streams where terrestrial invertebrates can be important, benthic stream invertebrates are the primary food base that supports drift feeding. Jowett (1990) showed that benthic invertebrate density is strongly correlated with trout abundance in New Zealand rivers. In addition, Weber et al. (2014) found that steelhead (anadromous rainbow trout) consumption rates were correlated with total drift biomass in tributaries of the Columbia River, Oregon. The density of benthic invertebrates over 3 mm indicates the availability of invertebrate food for medium to large fish (i.e. >200 mm) (Wankowski 1979; Hayes et al. 2000; Shearer et al. 2003).

If current speeds are too slow to support drift feeding, trout will browse for benthic invertebrates by picking them directly off the substratum, usually in pools or slow-flowing runs (Jonsson & Jonsson 2011). These habitats may be heavily silted and macrophytes, snails and chironomids can be abundant (Shupryt & Stelzer 2009). However, relative to drift feeding, benthic browsing is thought to be substantially less energy-efficient (Harvey & Railsback 2014). For example, Fausch et al. (1997) showed that numbers of Dolly Varden (Salvelinus malma) reduced by half (through emigration) when they were forced to benthic browse in a drift-blockage experiment in a natural stream.

We chose not to include a node that accounts for terrestrial-derived food for trout, because terrestrial insects provide only a small contribution to overall production in typical New Zealand lowland stream trout populations (McLennan & MacMillan 1984; Glova & Sagar 1991, Edwards & Huryn 1996). For example, Edwards & Huryn
(1995) showed that terrestrial invertebrates contributed as little as 5% to annual brown trout production in a relatively unmodified native grassland-pasture catchment.

Temperature plays an important role in determining if a fishery is limited by food. As discussed in the temperature limitation section above, energy requirements increase exponentially as temperatures increase above 14°C. Therefore, if food quantity was held equal, trout populations in warmer streams are more likely to be limited by food.

Trout density is another major factor influencing the likelihood of a trout population being limited by food availability (Railsback & Harvey 2011). If a trout population is limited by recruitment, there is less chance that food availability will limit growth and abundance because there will be less pressure on available food resources. In contrast, at high population densities interspecific competition limits production. This has been demonstrated by numerous authors who reported density-dependent growth or loss rates in juvenile trout populations (e.g. Crisp 1993; Hayes 1995; Jenkins et al. 1999; Grant & Imre 2005; Lobón-Cerviá et al. 2011).

1.6.1. Parent node(s) and input data

We chose five parent nodes to inform the food-limited node. These include: (1) the probability of forage fish presence (prey-fish node), (2) the density of benthic invertebrates (invertebrate-density node), (3) drift-feeding-opportunity node (which required a separate BBN sub-network), (4) the too-hot-limiting-factor node and (5) the recruitment-limited node.

Leathwick et al. (2005) modeled diadromous fish distributions for stream and river segments based on presence-absence data from the extensive New Zealand freshwater fish data base. Leathwick et al.’s species probability-of-occurrence
values can be considered a surrogate for abundance because both the likelihood of occurrence and abundance will be related to the strongest environmental predictors in the distribution model. These include distance from the sea, stream size, flow variation and summer temperatures. To maintain simplicity within our BBN, we have limited the prey-fish node to the likelihood of īnanga and smelt being present in the stream segment of interest—based on Leathwick et al.’s model predictions. These two common pelagic fish are more vulnerable to trout predation than cryptic, obligate benthic species such as common bullies.

To account for the potential for drift feeding we created a ‘drift feeding opportunity metric’. This metric required its own BBN sub-network, the specifications for which are detailed in the next section. For the invertebrate-density node, we opted to use benthic invertebrate density data obtained using the Stark et al. (2001) quantitative Surber sampling method. To account for the synergistic effect of high temperatures on food limitation, we linked the too-hot-limiting-factor node to the food-limited node. Likewise, to account for the negative effect of high fish densities on food availability we linked the recruitment- and food-limited nodes.

### 1.6.2. Data input categories

We suggest using the highest probability-of-occurrence for smelt or īnanga predicted with Leathwick et al.’s (2015) model (for the nearest REC segment to the LFA segment of interest) to inform the prey-fish node. Within the BBN, this node was defined as a continuous variable (i.e. no category breakpoints). To define the three categories within the invertebrate-density node, we used quantile classification of the Quinn & Hickey (1990) invertebrate data. We linked the drift-feeding-opportunity, too-hot and recruitment-limited nodes to the food-limited node as continuous variables.

To determine the conditional probabilities, we weighted the too-hot, recruitment-limited, prey-fish, drift-feeding-opportunity and invertebrate-density nodes 1, 1, 0.5, 0.5, and 0.5, respectively. We consider that high temperatures and high trout...
densities have an overriding positive influence on the probability that a trout population is limited by food relative to the other environmental-parent nodes.

1.7. Drift feeding opportunity

The opportunity for trout to drift feed in streams depends on invertebrate density and community composition. In general, Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa are larger and more prone to drift than other invertebrates. In particular, the mayfly Deleatidium spp. is common in the diets of drift-feeding New Zealand trout (Glova & Sagar 1991; McLennan & MacMillan 1984). Consequently, an invertebrate community with a high relative abundance of EPT taxa is thought to be more likely to support drift feeding (Shearer et al. 2003).

Identifying prey items in the drift is a cognitively demanding task for drift-feeding salmonids and the presence of algal fragments and other debris in the water column decrease feeding efficiency (O’Brien & Showalter 1993; Neuswanger et al. 2014). In addition, thick algal mats may make invertebrates less prone to enter the drift (Shearer et al. 2003; Jellyman & Harding 2016). Therefore, the presence of thick benthic algal mats should be considered when assessing drift-feeding potential.

As well as the biological factors influencing drift-feeding opportunity, water velocity and clarity factors also need to be considered. These factors required a separate sub-network to define which is described in the next section.

1.7.1. Parent node(s) and input data

Three parent nodes inform the drift-feeding-opportunity node including: (1) percentage abundance of EPT taxa (%EPT-abundance), (2) percentage algal cover (%-algal-cover and (3) the physical-drift-feeding-conditions (which required its own BBN sub-network to calculate).
The %EPT taxa abundance index is commonly used to indicate stream ecosystem health (Stark et al. 2001). As such, it is routine to extract this information from quantitative invertebrate data. A version of the %EPT taxa abundance index that excludes percentage abundance of hydroptilid caddisflies is more relevant for assessing invertebrate food potential for trout because it reduces the influence of small taxa (Collier 2009). A single benthic invertebrate sampling occasion following a stable flow period (of at least two weeks) during summer should be adequate to inform the %EPT-abundance node (Stark et al. 2001). However, the average values from longer-term invertebrate monitoring (e.g. 5 years of annual, summer sampling) would be preferable. We suggest visually estimating the combined percentage cover of filamentous algae and algal mats (>3 mm) within a stream reach to inform the %-algal-cover node. This should be done after a stable flow period (of two weeks or more) during mid-late summer (Biggs & Kilroy 2000).

1.7.2. Data input categories

To determine the three categories that define the %EPT-abundance node, we visually assessed the frequency distribution of Quinn & Hickey’s (1990) data for break points. These invertebrate data were collected as part of the 100 rivers survey (Jowett 1990). To define the binary categories for the %-algal-cover node, we chose <35% or ≥ 35 % composite algal mat (>3 mm thick) and filamentous algal cover. Matheson et al. (2016) recommends 35% composite algal mat cover as a maximum cover limit to protect trout habitat values in New Zealand rivers. We linked the physical-drift-feeding-conditions node (described below) to the drift-feeding-opportunity node as a continuous variable.

To determine conditional probabilities, we weighted the physical-drift-feeding-conditions and %EPT-abundance nodes twice that of the %-algal-cover node. This was to account for the relatively strong influence of physical variables and the food-base on drift feeding potential relative to the indirect effects of benthic algae on drift food quality and quantity. In addition, in most streams, nuisance algal growths occur only intermittently following extended periods of stable flow (Biggs et al. 1998).
1.8. Physical drift feeding conditions

Drift feeding by trout is dependent on local current velocities that can keep invertebrates in suspension and deliver a high rate of drift. On the other hand, velocities must be moderate enough to constrain fish swimming costs within ranges that return a net-positive energy intake rate. The ‘sweet spot’ velocity range occurs because although drift flux through a cross-sectional foraging area increases with velocity, the foraging area contracts, prey capture success declines and swimming costs increase exponentially (Hill & Grossman 1993; Grossman et al. 2002).

Invertebrate production occurs mainly in riffles and shallow runs (Keup 1988; Jowett & Richardson 1989; Brown & Brussock 1991) and pools offer deeper water in which adult trout can feed and find cover (Keup 1988; Baran et al. 1997; Heggenes et al. 1999; Armstrong et al. 2003). The transition zone between riffle or a fast run and pool (or a slow run) are preferred drift-feeding locations for trout, because they are close to the source of incoming drift and trout can find suitable feeding locations over the strong depth and velocity gradients. Therefore, the more riffle-pool sequences within a stream segment, the better the balance between food production and feeding areas for trout. The United States Environmental Protection Agency (USEPA) habitat assessment protocol for salmonids suggests that an ideal riffle to pool ratio is 1:1 (Barbour et al. 1999). This rule-of-thumb is thought to represent the ideal balance of adult trout habitat in the pools with food-producing and juvenile rearing habitat in the riffles.

Water clarity must also be sufficient for trout to see items in the drift. The maximum reaction distance for drift-feeding fish to detect prey is strongly affected by water clarity/turbidity (Sweka & Hartman 2001; Barrett et al. 1992; Hayes et al. 2016). When drift feeding, trout need enough time to react and intercept a prey item before it is carried downstream of their position (Hughes & Dill 1990; Railsback et al. 2009).
1.8.1. Parent node(s) and input data

The physical-variable drift feeding metric node has two parent nodes: (1) the relative percentage of riffles and/or fast runs to pools and/or slow runs (%-riffle-area) and (2) base-flow-turbidity.

The riffle and/or fast run: pool and/or slow run ratio for a river segment can be calculated using mesohabitat mapping data. This should be collected following visual estimate protocols (e.g. Barbour et al. 1999). These estimates should be undertaken throughout the stream segment of interest during flows that are close to the 7d-MALF. The riffle-fast run to pool-slow run ratio metric will approximate velocity diversity and the frequency of steep velocity gradients within a stream segment. These are areas that provide drift-feeding opportunities for trout of a range of year classes. We accept that defining fast runs and slow runs can be ambiguous. However, Barbour et al. (1999) provide good narrative descriptions which are adequate for segment scale assessments.

Long-term average turbidity (NTU) values of monthly base-flow (e.g. at least a 2-year record) are sufficient to inform the data requirements of the base-flow-turbidity node. Base-flow NTU spot measurements are taken at most regional authorities’ routine stream monitoring locations in New Zealand.

1.8.2. Data input categories

We defined the binary categories for the %-riffle-area node as within or outside a 30-70% riffle or fast run-area range. This range includes the riffle or fast run to pool ratio or slow run ratio of 1:1 which is presumed to be optimum for small stream salmonids (Barbour et al. 1999). We deliberately chose a broad range of percentage riffle or fast run areas to account for other factors that may influence the optimum amount of riffle (in relation to drift-feeding potential), such as the presence of instream structures creating patch-scale velocity gradients (Wheaton et al. 2017).
The prey reaction distance of drift-feeding salmonids reduces exponentially with increasing NTU values, reaching an asymptote at about 5 NTU (Gregory & Northcote 1993; Hayes et al. 2000; Railsback et al. 2009). We chose to interpret this continuous relationship according to the following three categories of NTU <1, 1-5, >5, as providing good, OK and poor water clarity for drift feeding, respectively. These NTU thresholds equate to approximate horizontal black-disk visual clarity readings of >1.2 m, 1.2-0.8 m and <0.8 m respectively—based on the NTU-visual range relationship presented in Davies-Colley & Close (1990).

### 1.9. Fish cover

Security cover is an essential component of fish habitat (Raleigh et al. 1986; Allouche 2002; Kratzer and Warren 2013). Any feature of a stream that obscures the bed from view can be considered fish cover. Common cover features include deep water (e.g. >1 m), macrophytes, bedrock, boulders and large, loose cobbles, undercut banks, overhanging vegetation and woody debris. Consequently, the presence and amounts of these features are often included in standard stream and salmonid habitat assessment protocols (e.g. Barbour et al. 1999).

#### 1.9.1. Parent nodes and input data

The fish-cover limiting-factor node has five parent nodes, (1) the percentage of water deeper than 1 m at base flow (%-depth >1 m), (2) %-overhanging-vegetation, (3) % undercut-banks, (4) %-macrophyte-cover and (5) %-woody-debris and/or boulders.

With the exception of percentage depth data, we suggest that the input data for all of the cover features should be based on visual estimates of percentage aerial cover within three 100 m stream reaches randomly located within a stream segment of interest (Holmes & Hayes 2011). The percent estimates of water deeper than 1 m can be obtained through a combination of depth measurements.
and visual estimates following the protocol detailed in Holmes & Hayes (2011). Estimates should be undertaken during summer low-flows.

1.9.2. Data input categories

The %-woody-debris and/or boulder node has two categories to represent the presence or absence of a substantial amount of wood or boulder cover in a reach. To distinguish between the two categories, we chose the nominal breakpoint of <5% or ≥ 5% of the wetted area of a stream segment. For the other fish-cover parent nodes, we assigned three broad categories that were defined by visually assessing the frequency distributions of habitat survey data from the CEP streams for breakpoints.

We weighted the cover features %-deep-water, %-woody-debris or boulders, % undercut-banks, %-overhanging-vegetation and %-macrophytes 1, 0.75, 0.75, 0.5 and 0.25, respectively, before calculating conditional probabilities. We assigned these weights by the ranked time-scale over which the cover features persist within a stream. We have assumed that relatively permanent cover features (e.g. undercut banks) are of more value to fish than features that can vary seasonally (e.g. macrophytes).

1.10. Water quality

Physicochemical water quality variables have the potential to limit trout production directly through physiological effects, or indirectly by altering ecosystem processes (e.g. high nitrate levels encouraging algal growth and causing subsequent changes in the invertebrate food web) (Quinn 2000). This section deals with direct effects of water quality on trout. Indirect effects on the food base are accounted for in separate sub-network branches of the BBN.

Trout need relatively good water quality to thrive in streams, particularly during the juvenile life history stages. Their temperature and dissolved oxygen (DO) requirements are well defined (Jonsson & Jonsson 2009). Nonetheless, determining the potential for adverse water quality effects is complicated by the
fact that water quality variables interact in non-additive ways to create more (or less) stressful conditions for biota (ANZECC 2000).

We focused on basic water quality parameters that we consider most likely to limit trout production in New Zealand lowland streams. These include temperature, DO, pH and nitrate-nitrogen (NO3-N). In lowland macrophyte-dominated streams, DO and pH can fluctuate severely (Burrell et al. 2013), which can result in stressful condition for trout (Davies-Colley et al. 2013). Nitrate-nitrogen (NO3-N) is a pervasive non-point source pollutant in catchments dominated by intensive agriculture (Quinn 2000) and NO3-N concentration in some agricultural catchments may induce chronic effects on early life-history-stage rainbow trout. To our knowledge, there is no research available to determine NO3-N levels that could induce chronic effects brown trout eggs or alevins (Burdon & Taylor 2000). Suspended sediment (SS) can also directly stress trout, for example by clogging gill membranes (Railsback et al. 2009). However, behavioural effects (e.g. impaired drift-feeding ability) can be expected to occur for salmonids at low suspended sediment levels relative to the levels required to cause acute effects (i.e. <5 NTU vs. >20 NTU) (Davies-Colley et al. 2015). Therefore, we did not include a SS node in the water quality BBN sub-net. Nevertheless, we do address the indirect effect of SS (using turbidity as a surrogate) on feeding efficiency in the drift-feeding opportunity sub-network.

1.10.1. Parent nodes and input data

Four parent nodes are linked to the water-quality-limited node: (1) DO, (2) pH, (3) nitrate-nitrogen (NO3-N) and 4) the too-hot limiting-factor node.

Dissolved oxygen and pH can be measured using standard water quality probes. Spot DO measurements should be taken in mid to late summer during base flows following at least a two-week period of stable flows to allow any algal growths to occur that may increase respiration and decrease DO at times (Davies-Colley et al. 2015).
In addition, measurements should be taken close to dawn when DO concentration is lowest due to stream respiration and no photosynthesis overnight. Continuous oxygen loggers are becoming cheaper and more commonplace for routine monitoring. If deployed over a summer low-flow period, the minimum DO concentrations from these data should be used if available. By contrast, pH is best measured mid-afternoon during summer base-flow when photosynthesis peaks (Davies-Colley et al. 2011). Water samples for nitrate analysis should be taken in early spring near spawning and fry rearing habitat. This is where emerging fry will be most vulnerable to nitrate toxicity and nitrate levels are at their highest owing to saturated ground and winter-farming practices such as break-feeding (Quinn 2000).

### 1.10.2. Data input categories

Davies-Colley et al. (2013) provided a set of proposed DO condition bands to inform the NOF initiative. These condition bands are intended to provide DO limits which provide various degrees of protection for the ‘life supporting capacity’ of freshwater ecosystems. The more stringent DO condition bands were based on DO concentration tolerance of rainbow trout. These concentrations are likely to be equivalent to tolerance thresholds for brown trout. Accordingly, we chose Davies-Colley et al.’s four suggested DO ‘condition band’ ranges to define our DO node category breakpoints.

The longstanding ANZECC (2000) guidelines define the suitable pH range for salmonids as being between 6 and 9. We converted this range to a binary variable (within or outside 6-9 pH) to define the two pH node categories.

Research is required to determine accurate nitrate toxicity thresholds for brown trout. Hickey (2013) reviewed nitrate toxicity literature for freshwater fish and invertebrates and proposed NOF protection bands for New Zealand streams. These were based in part on estimated chronic NO3-N toxicity thresholds for salmonids. The no-effect concentration (NOEC) of NO3-N for rainbow trout is around 2.4 mg/l. This value was determined from the geometric mean of just two
NOEC values derived from experiments undertaken by Kincheloe et al. (1979). Subsequent long-term nitrate toxicity tests by Hickey (2013) found that a New Zealand strain of rainbow trout was nine times less susceptible to nitrate than the strain studied by Kincheloe et al. (1979). Nevertheless, to account for considerable uncertainty surrounding NO3-N toxicity for salmonids, and the lack of direct NO3-N chronic toxicity data for brown trout, we chose the (conservative) proposed NOF nitrate condition bands to inform our nitrate toxicity category thresholds (Hickey 2013). These are based on NOEC for lake trout (*Salvelinus namaycush*), which is the most nitrate-sensitive fish species present in New Zealand fresh waters.

We weighted all water quality variables equally when determining conditional probabilities. However, to account for the synergistic effect of high temperatures on water quality stress (see e.g. Bruder et al. 2017), we multiplied the DO, pH and nitrate variables by the too-hot limiting-factor node if it was within a range that would impair trout metabolism.

### 1.11. Social limitation to fishery usage

Humans are an essential component of a fishery. Consequently, poor aesthetic values or human health risk factors potentially limit angler usage. When selecting the parent nodes for this BBN sub-network, we chose to focus on instream values for which pre-existing numerical guidelines exist in relation to angling values.

Counts of the faecal bacteria *E.coli* in water samples are commonly used to indicate the health risk of undertaking ‘secondary contact’ recreational actives such as angling. In addition, percentage algal and macrophyte cover guidelines for trout angling aesthetics are available from Biggs & Kilroy (2000), recently modified by Matheson et al. (2016).
1.11.1. Parent nodes and input data

The usage-limited child node has three parent nodes: (1) contact recreation status based on E.coli counts (rec-status), (2) %-algal-cover and (3) %-macrophyte-cover.

Water quality testing for E.coli is routinely undertaken by regional authorities at the network of sites associated with the New Zealand State of the Environment monitoring programme. However, if long-term E.coli data do not exist for a stream, then E.coli samples should be collected monthly during low-to-moderate flows when anglers are likely to be fishing (MFE 2002). To inform the %-algal-cover and %-macrophyte-cover nodes, we suggest obtaining visual estimates of percentage cover from three 100 m reaches that are randomly located within the focal stream segment (as detailed in Holmes & Hayes 2011).

1.11.2. Data input categories

There is considerable ongoing debate over the appropriate level of health risk imparted by different concentrations of E.coli in waterway samples (McBride 2012). Therefore, rather than using counts to inform the category breakpoints for the rec-status node, we opted for the contact recreational status of the waterway as defined by contemporary risk profile calculation standards (e.g. MFE 2002). These include the categories ‘no contact’, ‘secondary contact’ (i.e. contact without submerging head, e.g. wading) through to primary contact (i.e. swimming / full immersion).

Matheson et al. (2016) provide an interim guideline of <35% composite cover of filamentous algae and algal mats (>3 mm thick) to protect angling aesthetic values. We converted this threshold to binary categories of <35% or ≥ 35% to define the percent algal-cover node breakpoints.

We weighted contact recreational status, percent algal cover and percent macrophyte cover 1, 0.66 and 0.33, respectively, to reflect our expert opinion of anglers’ attitudes towards the severity of these potential issues within a fishery.
1.12. DIAGNOSTIC-TEST NODES

The limiting-factor nodes are the parent nodes to a set of fishery diagnostic metric nodes. These nodes include six standard metrics used by researchers and managers to monitor fisheries. We have selected metrics that vary in sophistication and expense to allow for different monitoring budgets. Our diagnostic metrics include:

1) January–March Relative Weight
2) Observed vs. expected growth rate
3) Young-of-the-year density (fish / m²)
4) Biomass (g / m²) of trout ≥ 200 mm
5) National Angler Survey (NAS) usage statistics
6) Enjoyment / importance survey scores.

The reasons for including these metrics, as well as the methods to determine the category breakpoints and conditional probabilities, are discussed below.

1.12.1. Relative weight (Wr)

One of the simplest and most widely used metrics for determining individual and fish population health is condition factor (K)—i.e. the length to weight ratio of a fish (Fulton 1902; Ricker 1975). However, more recently, relative weight (Wr) has been proposed as a more versatile measurement of fish condition (Murphy et al. 1991, Blackwell et al. 2000). This is because this metric does not assume isometric growth and therefore can be used to compare individuals with different lengths (Blackwell et al. 2000).

Relative weight is calculated using the following formula:

\[ Wr = \left( \frac{W}{Ws} \right) \times 100 \]
where \( W = \) actual fish weight, and \( W_s = \) a ‘standard weight’ for fish of the same length. Milewski & Brown (1994) determined \( W_s \) for stream brown trout from populations across the United States and Canada (\( n = 10,673 \)) using the following formula:

\[
\log W_s = -5.422 + 3.194 \times \log TL
\]

where \( W_s \) is in grams and TL (total length) is in millimetres. We see no reason why this formula will not be applicable to New Zealand trout populations.

The effects of spawning can substantially affect \( W_r \), even in rivers with abundant food. Therefore, for the BBN, \( W_r \) should be estimated for fish sampled in mid-summer to early autumn (January–March in New Zealand) when fish ought to be in peak condition. We suggest that an unbiased sample (i.e. not obtained by angling) of at least 20 fish is needed to generate average \( W_r \) input values (Blackwell et al. 2000). Preferably, sample collection should be undertaken over multiple years.

We linked the \( W_r \) node to the food-limited node because the most likely explanation for poor average condition in a population is food limitation. We determined two categories for the \( W_r \) node either above or below a \( W_r \) value of 85. Relative weight values below or above this value are considered to indicate poor or adequate condition, respectively (Blackwell et al. 2000).

### 1.12.2. Observed vs. expected growth rates

Bioenergetics growth models can be used to indicate whether fish populations are food limited by comparing observed growth rate with predicted growth rate assuming maximum rations (expected growth) (Hewett & Johnson 1992; Hansen et al. 1993; Railsback & Rose 1999). Predicted growth is varied until it matches observed growth and the resulting value (\( C/C_{\text{max}} \), where \( C = \) consumption and \( C_{\text{max}} \) is maximum consumption based on invertebrate or fish diet or a
combination of both) provides an indication of the degree of food limitation. For stream trout growth modeling, we suggest modeling an invertebrate diet. Observed growth rate can be estimated for multiple years from a size-at-age sample, with age being estimated from otoliths or, for juvenile fish, from scales. A sample of at least 20 fish is required for growth estimation. Detailed methods for applying bioenergetics trout growth modeling are given in Hayes (2013). Because growth modeling relates directly to food limitation, we linked it with the food-limited node as a continuous variable in the BBN.

1.12.3. Biomass of brown trout over 200 mm

The maritime climate and temperate latitude of New Zealand streams provide close to ideal temperatures for trout growth. Growth is generally rapid in lowland-mid altitude streams and rivers, such that by the end of their second year, brown trout usually reach about 200 mm (e.g. Hayes et al. 2000). Therefore, the biomass of fish >200 mm is an indicator of the potential pool of mature fish approaching spawning age. We chose biomass and not density because it better represents the productivity of a fishery (Melstrom et al. 2015). Various methods are available for estimating medium to large trout biomasses, including electric fishing and visual surveys by snorkel divers (Johnson et al. 2007). However, water clarity is often insufficient to visually survey lowland streams. Therefore, we suggest using quantitative electric fishing (multiple pass depletion count method) to provide input data for this node. Standard protocols for quantitative electric fishing are given in Johnson et al. (2007).

We determined the categories for our biomass breakpoints (in g / m²) by visually assessing the frequency distribution of trout biomass for break points across the range present in the CEP data. The ‘trout >200 mm biomass’ node was linked to the food-limited, water-quality-limited, low-flow-limited and fish-cover-limited nodes. We used a conceptual model described in Chapter 2 methodology (Section
2.3, Figure 4A) to determine the conditional probabilities for the trout >200 mm biomass node.

1.12.4. Young-of-the-year density

Recruitment potential can be measured directly from the density of young-of-the-year (YoY) trout estimated from quantitative electric fishing. Data should be gathered in mid summer to autumn (January–March in New Zealand) to determine densities after the population bottleneck that occurs following spring emergence (Jensen & Johnsen 1999). Long-term annual sampling is recommended in order to account for environmental stochasticity (e.g. 10 years) (Frank et al. 2011).

We linked the YoY-density node to the flood-limited, recruitment-limited and water-quality-limited nodes. As discussed in Sections 1.2, 1.5 and 1.10 (respectively), these variables can have severe and disproportionate impacts on juvenile trout survival relative to adult trout. We determined the juvenile trout density category values by visually assessing the frequency distribution of YoY densities within the CEP data for breakpoints. We used a conceptual model described in Chapter 2 methodology (Section 2.3, Figure 4B) to determine the conditional probabilities for the YoY density node.

1.12.5. National angling usage survey statistics

The National Angling Usage Survey (NAS) has been undertaken approximately every five to seven years since 1994 (e.g. Unwin 2016). This resource gives a reliable estimate of the usage of the more popular fisheries in New Zealand, and the survey results can inform whether a fishery is likely to be unpopular. Accordingly, we linked this node to the usage-limited node. We set the lower usage-breakpoint at <100 angler days per year to indicate ‘unpopular fisheries’ based on the nominal value for ‘low-usage’ rivers suggested by Unwin (2013). We chose the upper categories of 100 - 500 and >500 angler days based on the NAS usage rates in moderately popular and popular New Zealand wadeable-stream fisheries. To derive these values, we considered usage rates of ‘moderately popular’ and ‘popular’ New Zealand wadeable stream fisheries based on the lead
authors’ expert opinion. This was guided by how often the streams assessed are mentioned in popular fishing documents and fishing guides (e.g. Kent 2009).

1.12.6. **Enjoyment / importance score**

As part of the 2011 / 2012 NAS, Unwin (2013) conducted an investigation of the ‘values of New Zealand angling rivers’. He assessed the quality of rivers and streams by phone interviews of anglers, asking them to score nine attributes for each river they had fished. These attributes included: ‘close to home’, ‘close to holiday home’, ‘easy access to river’, ‘plenty of fishable water’, ‘scenic beauty’, ‘wilderness feeling’, ‘angling challenge’, ‘expect good catch rate’, and ‘chance to catch trophy fish’. Unwin (2013) combined the attribute scores to generate overall average ‘enjoyment / importance’ scores that give a gross indication of the relative quality of the fisheries. We included an enjoyment / importance node in the set of diagnostic fishery metrics. The categories for this node were binary, ‘above or below’ the national overall mean enjoyment / importance score of 2.36 (out of a possible 5). We linked the enjoyment / importance node to the usage-limited node. If no data exist for a given catchment, anglers that are familiar with the waterway could be surveyed using Unwin’s method.
Appendix 2. In stream field habitat assessment guide and field sheet.

**In-stream assessment field guide**

Peg out a 20 m sub-reach at the start (0 m), midpoint (10 m) and endpoint (20 m). Work from downstream to upstream. Measure the wetted width at the start and end of each 20 m sub-reach (if the width changes by more than half of one of these measurements then take a midpoint width measurement as well). Record the GPS position (NZ map grid) on the true right bank (right-hand side looking downstream) at the start and end of each 20 m sub-reach. Fill out one in-stream field form for each 20 m sub-reach.

Follow steps 1 – 7 to fill out the in-stream field form:

1. Record the percent area cover of the mesohabitat type(s) present in the sub-reach - fast run, slow run, riffle and pool (See Table 1).
2. Record the percent area of the following depth classes <0.3m, 0.3-0.5, 0.5-1m and >1m deep - split the percent area estimates of these depth categories into 10m sections of the sub-reach as indicated on the field sheet.
3. Record the percent cover of the dominant substrate size classes within the areas of visible stream bed (i.e. stream bed that is not obscured by macrophytes/weed), see Table 2 for sediment codes, or ‘Sediment assessment method 1’ in the sediment assessment methods protocol. Estimate the percent cover of any macrophyte/weed beds. Estimate the percent cover of any thick (>3 mm) algal mats or green filamentous algal mats greater than 2 mm long (see Table 2).
4. Measure or estimate the percent cover, or area in m² (depending in cover type), of any in-stream fish cover (Table 3).
5. If a shallow run (<1 m deep) or pool-tailout is present in the 20m sub-reach then undertake a “shuffle test” assessment. See ‘Sediment assessment method 5’ in the sediment assessment methods protocol.
6. Record the residual pool depth – measure the deepest point and shallowest point at the riffle crest downstream (see additional residual pool depth protocol sheet).
7. Check that you have completed all the fields in the field sheet.
## Mesohabitat classifications

### Table 1. Descriptions and codes for the various mesohabitat units and habitat sub-units.

<table>
<thead>
<tr>
<th>Mesohabitat type</th>
<th>Habitat sub-unit</th>
<th>Description</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Riffle</td>
<td>Shallow to moderate depth, moderate to fast water velocity with mixed currents, surface rippled but unbroken.</td>
<td>(hab) LGR</td>
<td></td>
</tr>
<tr>
<td>Run</td>
<td>Fast run</td>
<td>Habitat in between that of riffle and pool, moderate to fast water velocity, slightly variable current surface smooth and/or rippled.</td>
<td>(hab) FRun</td>
</tr>
<tr>
<td></td>
<td>Slow run</td>
<td>Habitat in between that of riffle and pool, moderate depth and moderate to slow water velocity, uniform and/or slightly variable current surface unbroken smooth and rippled sections.</td>
<td>(hab) SRun</td>
</tr>
<tr>
<td>Pool</td>
<td>Deep, slow-flowing with a smooth water surface, usually where the stream widens or deepens.</td>
<td>(pool)</td>
<td></td>
</tr>
</tbody>
</table>

## Sediment classifications

### Table 2. Descriptions and codes for the different stream-bed features that should be recorded

<table>
<thead>
<tr>
<th>Stream bed characteristic</th>
<th>Description</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Estimate the percentage of the visible stream bed dominated by the main substrate sizes</td>
<td>mud/silt/sand - FS (&lt;2mm), fine gravel - FG (2-32mm), coarse gravel - CG (&gt;32-64), small cobble - SC (&gt;64-128mm), large cobble - LC (&gt;128-256), boulder - B (&gt;256mm) and bedrock - BR (continuous)</td>
<td>(sed) %</td>
</tr>
<tr>
<td>Estimate the percentage coverage of any macrophytes</td>
<td>Record % coverage of conspicuous macrophytes (stream weed), record type if known.</td>
<td>(weed) %</td>
</tr>
<tr>
<td>Estimate the percentage coverage of any algal mats</td>
<td>Record % coverage of conspicuous algal mats (&gt;3mm) or green filamentous algae greater than 2mm.</td>
<td>(algal) %</td>
</tr>
</tbody>
</table>
Fish cover classifications

Table 3. Descriptions and codes for the different cover features that should be recorded

<table>
<thead>
<tr>
<th>Cover type</th>
<th>Description</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Turbulence/broken water (obsured stream bed)</td>
<td>Record the presence of turbulent water cover if the stream bed is obscured and &gt;0.3m deep.</td>
<td>(cov) Turb</td>
</tr>
<tr>
<td>Bed Rock/large boulders</td>
<td>Record the presence and estimate the size of any large boulders &gt;0.5m or bed rock.</td>
<td>(cov) Rock</td>
</tr>
<tr>
<td>Large woody debris</td>
<td>Record the presence and estimate the size of woody debris (&gt;1m*0.3m) only include wood debris (or the part) that is within the wetted channel.</td>
<td>(cov) WD</td>
</tr>
<tr>
<td>Submerged branches</td>
<td>Record the presence and estimate the size of matted submerge branches (&gt;1m*0.3m) only include branches (or the part) that is within the wetted channel.</td>
<td>(cov) SB</td>
</tr>
<tr>
<td>Man-made cover</td>
<td>Record the presence and estimate the size of any man-made structures only include the structure (or the part) that is within the wetted channel; (record any relevant notes on the type of man-made cover e.g. bridge pylons, flood protection works).</td>
<td>(cov) Man</td>
</tr>
<tr>
<td>Undercut banks</td>
<td>Record the presence and annotate the length of any undercut banks using the undercut-depth categories 0-0.3m, 0.3-0.5m, 0.5-1m and &gt;1m.</td>
<td>(cov) Undercut</td>
</tr>
<tr>
<td>Overhanging / emergent Vegetation</td>
<td>Record the presence and annotate the length of any overhanging vegetation only record vegetation that is touching to water surface or obscuring the stream bed from view use the overhanging categories 0-0.3m, 0.3-0.5m, 0.5-1m and &gt;1m. Estimate the percentage of the stream bed that is obscured from view.</td>
<td>(cov) OV</td>
</tr>
</tbody>
</table>
### Field sheet

<table>
<thead>
<tr>
<th>Stream:</th>
<th>Site (20m sub-reach) number:</th>
<th>Date:</th>
<th>Assessor team:</th>
</tr>
</thead>
<tbody>
<tr>
<td>D/S Width (m):</td>
<td>D/S gps (true right):</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Midpoint width (m):</td>
<td></td>
<td>e</td>
<td>n</td>
</tr>
<tr>
<td>U/S, Width (m):</td>
<td>U/S gps (true right):</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

% Mesohabitat types for 20m sub-reach:

<table>
<thead>
<tr>
<th>Riffle</th>
<th>Slow run</th>
<th>Fast run</th>
<th>Pool</th>
</tr>
</thead>
</table>

% Area depths for 0m to 10m mark:

<table>
<thead>
<tr>
<th>0 – 0.3 m deep</th>
<th>0.3 – 0.5 m deep</th>
<th>0.5 – 1 m deep</th>
<th>1 m + deep</th>
</tr>
</thead>
</table>

% Area depths for 10m to 20m mark:

<table>
<thead>
<tr>
<th>0 – 0.3 m deep</th>
<th>0.3 – 0.5 m deep</th>
<th>0.5 – 1 m deep</th>
<th>1 m + deep</th>
</tr>
</thead>
</table>

% Substrate Type for 20m sub-reach:

<table>
<thead>
<tr>
<th>Fine sed (&lt;2mm)</th>
<th>Fine gravel (2 – 32mm)</th>
<th>Coarse gravel (32-64mm)</th>
<th>Small cobble (64-128mm)</th>
<th>Large cobble (128-256mm)</th>
<th>Boulder (&gt;256mm)</th>
<th>Bed rock (continuous)</th>
</tr>
</thead>
</table>

% Weed/macrophyte cover

% Algal cover (include only filamentous or thick algal mat cover >3mm)

Shuffle test score (1-5 scale): (in run or pool-tail/glide only)

Fish cover:

**Undercut bank - True Right bank** (linear m of bank edge length – i.e. maximum of 20m)

<table>
<thead>
<tr>
<th>0 – 0.3 m deep</th>
<th>0.3 – 0.5 m deep</th>
<th>0.5 – 1 m deep</th>
<th>1 m + deep</th>
</tr>
</thead>
</table>

**Undercut bank - True Left bank** (linear m of bank edge length – i.e. maximum of 20m)

<table>
<thead>
<tr>
<th>0 – 0.3 m deep</th>
<th>0.3 – 0.5 m deep</th>
<th>0.5 – 1 m deep</th>
<th>1 m + deep</th>
</tr>
</thead>
</table>

**Overhanging veg - True Right bank** (linear m of bank edge – i.e. maximum of 20m)

<table>
<thead>
<tr>
<th>0 – 0.3 m</th>
<th>0.3 – 0.5 m</th>
<th>0.5 – 1 m</th>
<th>1 m +</th>
</tr>
</thead>
</table>

**Overhanging veg - True Left bank** (linear m of bank edge – i.e. maximum of 20m)

<table>
<thead>
<tr>
<th>0 – 0.3 m</th>
<th>0.3 – 0.5 m</th>
<th>0.5 – 1 m</th>
<th>1 m +</th>
</tr>
</thead>
</table>

**Woody debris (m²)** (include items >0.5m*0.3m)

**Submerged branches (m²)** include items (>1m*0.3m)

**Turbulence (m²)** (Include if the stream bed is obscured and depth is >0.3m.)

**Manmade cover (m²)** (e.g. rip-rap, bridges, old tires)

Residual Pool depth (m):

<table>
<thead>
<tr>
<th>Deepest point depth</th>
<th>D/S Hydraulic control depth</th>
<th>Comments</th>
<th>Photo taken?</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

*Photo taken?*

*All sheet fields completed?*
Appendix 3. Riparian field habitat assessment guide and field sheet.

**Stage one: riparian, land-use and contaminant source assessment field guide.**

For hardcopy photographs, **when possible trace around the habitat feature if it can be seen on the photo** and attach the relevant code (see Tables 1 and 2). If the feature is too small to be seen on the photo, then place a point at its location and estimate its size ($m^2$) and/or length (m).

Note the land-use type: crop, sheep/beef cattle, dairy or other (specify) - record land-use type only at the start of the assessment or when it changes.

Follow steps 1 – 7.

8. Trace along the wetted edge of the stream
9. Trace around and/or estimate the average width of the (fenced) riparian management zone. Record the level of stock access within the riparian zone – no access, fenced with periodic/partial access, open access
10. Record the vegetation types within the riparian management zone (see veg. codes in Table 1)
11. Trace around or estimate the area ($m^2$) of vegetation within 30 m of the stream edge - record vegetation type (see veg. codes in Table 1) estimate height of any trees to the nearest 5 m.
12. Note, trace around or estimate the area ($m^2$) of any vegetation overhanging the stream edge (see Table 1)
13. Trace along the length, or estimate the area (if it is too small to be seen on the aerial photograph) of any significant potential sources of sediment (i.e. bank slumping or stock pugging (see Table 2)
14. Note any drains or tributaries and label type (eg: small tributary, open drain, mole/tile). Record if the tributary/drain is fenced or not. Record other land-use features described in Table 2.

See Figure 1 for an example of a completed section of mapped riparian habitat.
Figure 1. An example of a completed section of the riparian survey (one of the five aerial photos used to survey this segment in the Waikakahi Stream, South Canterbury).
**Riparian features**

Table 1. Descriptions and codes for the various riparian features that should be recorded

<table>
<thead>
<tr>
<th>Riparian feature</th>
<th>Riparian sub-feature</th>
<th>Description</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Trees present in segment</td>
<td>n/a</td>
<td>Note or trace around any solitary large trees (e.g. willows) or prominent vegetation features like large flax stands that are present along the stream bank. Note the tree type (if known) estimate the height of trees.</td>
<td>(rip) Tree (e.g. willow)</td>
</tr>
<tr>
<td>Swamp/marsh grasses/herbs</td>
<td>Exotic</td>
<td>Trace around or estimate the percent area of exotic swamp/wetland herbs/grasses and/or large areas of emergent vegetation, note any of riparian seepage areas (where soils are waterlogged and have some wetland plants present).</td>
<td>(rip) E Swamp</td>
</tr>
<tr>
<td></td>
<td>Mixed Native and Exotic</td>
<td>Trace around or estimate the percent area of mixed exotic and native swamp/wetland grasses and herbs and/or large areas of emergent vegetation.</td>
<td>(rip) Mix Swamp</td>
</tr>
<tr>
<td></td>
<td>Native</td>
<td>Trace around or estimate the percent area of native swamp/wetland grasses and herbs and/or large areas of emergent vegetation.</td>
<td>(rip) N Swamp</td>
</tr>
<tr>
<td>Grasses/herbs:</td>
<td>Exotic</td>
<td>Record the presence of exotic grasses.</td>
<td>(rip) E Grass</td>
</tr>
<tr>
<td></td>
<td>Mixed Native and Exotic</td>
<td>Record the presence of exotic grasses mixed with native tussocks and/or sedge grasses.</td>
<td>(rip) Mix Grass</td>
</tr>
<tr>
<td></td>
<td>Native</td>
<td>Record the presence of native tussocks, flax and/or sedge grasses.</td>
<td>(rip) N Grass</td>
</tr>
<tr>
<td>Shrub (&lt;2 m high)</td>
<td>Exotic</td>
<td>Record the presence of exotic shrubs e.g. gorse and broom.</td>
<td>(rip) E Shrub</td>
</tr>
<tr>
<td></td>
<td>Mixed Native and Exotic</td>
<td>Record the presence of mixed Exotic shrubs, e.g. gorse, broom, blackberry occurring with native shrubs such as matagouri or manuka.</td>
<td>(rip) Mix Shub</td>
</tr>
<tr>
<td></td>
<td>Native</td>
<td>Record the presence of native shrubs (e.g. matagouri or manuka).</td>
<td>(rip) N Shrub</td>
</tr>
<tr>
<td>Tree</td>
<td>Exotic</td>
<td>Record the presence of exotic tree stands, record type if known (e.g. pine, gum, or willows. Estimate height to the nearest 5m.</td>
<td>(rip) E Tree</td>
</tr>
</tbody>
</table>
Mixed Native and Exotic

<table>
<thead>
<tr>
<th>Land-use feature</th>
<th>Description</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mixed Native and Exotic</td>
<td>Record the presence of mixture of mature native and exotic trees e.g. willow and cabbage trees. Estimate height to the nearest 5m.</td>
<td>(rip) Mix Tree</td>
</tr>
<tr>
<td>Native</td>
<td>Record the presence of mature native bush. Estimate height.</td>
<td>(rip) N Tree</td>
</tr>
<tr>
<td>Mixed mature canopies</td>
<td>Exotic</td>
<td>Record the presence of mature riparian zone with a mixed canopy of exotic grasses, shrubs and trees.</td>
</tr>
<tr>
<td>Mixed Native and Exotic</td>
<td>Record the presence of mature riparian zone with a mixed canopy of exotic and native grasses and or grasses, shrubs and trees.</td>
<td>(rip) Mix canopy</td>
</tr>
<tr>
<td>Native</td>
<td>Record the presence of mature riparian zone with a mixed canopy of native grasses and or grasses, shrubs and trees.</td>
<td>(rip) N canopy</td>
</tr>
</tbody>
</table>

### Land use features

Table 2. Descriptions and codes for various potential sediment and contaminant sources

<table>
<thead>
<tr>
<th>Land-use feature</th>
<th>Description</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>grazed or un-grazed</td>
<td>Note if the riparian management zone is freely grazed by stock, has partial or periodic grazing or stock have no access.</td>
<td>(rip) Graze (no access, partial graze, grazed)</td>
</tr>
<tr>
<td>Bank slumping</td>
<td>Record linear length (m) of bank erosion - defined as visible bank scars with exposed earth (annotate the location on the aerial photograph)</td>
<td>(land) Slump</td>
</tr>
<tr>
<td>Stock access</td>
<td>Record presence/absence of fencing (yes/no?), (annotate any fence gaps or natural stock barriers).</td>
<td>(land) Fence (y or n)</td>
</tr>
<tr>
<td>Stock barriers where fencing does not exist</td>
<td>Record the presence of cliffs or significant stands of dense vegetation.</td>
<td>(land) Stock B (y or n)</td>
</tr>
<tr>
<td>Stock crossings</td>
<td>Note areas where stock can/have been driven through the stream.</td>
<td>(land) Stockcross</td>
</tr>
<tr>
<td>Stock pugging</td>
<td>Note areas where stock have accessed the stream edge and caused damage to the bank. Record the area of damage (m²) or if present as a narrow strip (i.e. &lt;2m wide) record the linear length (m).</td>
<td>(land) Stockpug</td>
</tr>
<tr>
<td>Laneways</td>
<td>Note any cattle tracks adjacent to stream.</td>
<td>(land) Laneway</td>
</tr>
<tr>
<td>Tributaries/drainage</td>
<td>Record where any tributary drains or ditches that drain paddocks</td>
<td>(land) Trib</td>
</tr>
<tr>
<td>Standoff pads</td>
<td>Note the presence of any sacrificial paddocks where stock are kept for extended periods of time and supplied food.</td>
<td>(land) Standpad</td>
</tr>
</tbody>
</table>

enter the stream. Record if stock have access to the drains and if the drain has obvious pugging or turbidity and is likely to contribute sediment to the stream.