Effects of disturbance and nutrient regimes on freshwater invertebrate community structure

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Abstract

Freshwater ecosystems globally are under threat from anthropogenically driven impacts including water abstraction for drinking and agriculture, exotic species invasion, eutrophication, channelization and destruction or modification of habitat. In New Zealand, eutrophication from nutrient enrichment is one of the most pervasive and detrimental impacts. High nutrient levels in waterways is detrimental not only to the species that inhabit them, but can also affect drinking and irrigation water for humans and result in loss of recreational and aesthetic values. Excess nitrogen and phosphorus can cause large amounts of periphyton to grow which in turn can impact water quality and the community composition of stream macroinvertebrates. Macroinvertebrate communities are also strongly affected by floods. More or less frequent flooding can cause changes in composition of stream invertebrate communities. Streams are usually affected by multiple stressors but the effect of those stressors are often considered in isolation for management. As macroinvertebrates are often used as indicators of ecosystem health, it is important to assess how different stressors interact and how these impacts those communities.

In this study, macroinvertebrate communities in four Taranaki streams were sampled to assess the interactive effects of nutrient enrichment and flood regime. Nutrient enrichment resulted in invertebrate communities changing markedly between upstream and downstream sites. All four streams had a similar composition at the upstream sites whereas downstream sites in most streams were very different. In two of the streams with lower disturbance regimes, nutrients were the most important driver of invertebrate community composition. In the two streams with a higher disturbance regime, the invertebrate communities were more similar between upstream and downstream sites indicating that flooding was overriding the effects of nutrient enrichment as the most important driver of community composition.

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E rere kau mai te awa nui nei
Mai i te kāhui maunga ki Tangaroa
Ko au te awa
Ko te awa ko au.

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Chapter 1: The interaction between nutrients and flood regime in affecting stream macroinvertebrate communities

Global freshwater issues

Freshwater makes up 0.01% of the world's water, and covers approximately 0.8% of the Earth's surface yet this tiny fraction supports over 100,000 species – roughly 6% of the total species on Earth (Dudgeon et al., 2006). This freshwater and its associated biodiversity constitutes a precious natural resource not only for its intrinsic value, but also for it's economic, cultural, aesthetic, scientific and educational values. Yet freshwater ecosystems are experiencing far greater declines in biodiversity than the most impacted terrestrial ecosystems. The health of these freshwater systems is essential to the survival of humans and the many other species that rely on its life supporting qualities.

Humans now exert considerable influence on almost every freshwater ecosystem on the planet. The most prominent anthropogenically driven impacts on freshwater quality and biodiversity are abstraction, water pollution, exotic species introductions or invasions, flow modification and habitat destruction and degradation. These impacts manifest themselves mainly in the dramatic alteration of nutrient fluxes between the landscape and the receiving waters. These altered regimes have serious deleterious effects on the quality of surface waters across the globe. In addition to this, the growing human population has meant growing water demand for agriculture, drinking water, sanitation and hygiene (Abell, 2002).

Another wide-ranging, anthropogenically driven issue affecting freshwater is climate change. Rapid human population increase over the last 150 years has meant an increase in greenhouse gas emissions from agriculture, industry and transportation, resulting in unprecedented increases in temperatures on Earth. This increase in average temperatures has affected precipitation patterns (Stocker, 2014). This means that the occurrence of extreme weather events such as floods and drought will be altered or become more frequent. Karl and Knight (1998) found that arid regions of the world were becoming drier while middle and northern latitudes were becoming wetter. There has been a disproportionate increase in heavy precipitation events and corresponding flooding in these areas. An extreme flood can be defined as a flood with a 100 year return interval or floods with a 1% exceedance probability (Holmes Jr & Dinicola, 2010). Nottage et al. (2012) predict that in New Zealand, climate change will affect catchments differently. For example, rivers fed by snow melt are predicted to have increased flows and groundwater-fed rivers to have decreased flows. The seasonal timing of these extreme floods may also be altered by climate change resulting in high flow periods which occur out of phase with flow related life histories such as spawning and migration (Delong, Thorp, Greenwood, & Miller, 2001). Change in river flows over time are determined by changes in temperature and most importantly, precipitation (Chiew, 2006). Extreme flooding can alter river geomorphology and affect the available habitat over a longer period of time than just during high flow events (Death et al., 2015). The influence that flooding has upon the morphology of a river depends on a number of factors. Sediment load, size, river stability and sinuosity are important factors contributing to how much morphological change a flood may cause (Brunsden, 2001). While there is some uncertainty about the effects that extreme floods can have upon river geomorphology, some possible effects may be predicted. Channels may be widened, sediment deposited on river margins and channels will straighten and river bed erosion can result in altered depth profiles (R. G. Death, Fuller, & Macklin, 2015). Increased episodes of extreme flooding resulting in geomorphological changes can exacerbate anthropogenic stressors. Intense precipitation affects the rate at which materials such as sediment, nutrients or toxic chemicals are flushed into rivers in run off (Kundzewicz et al., 2008). An increase in weathering or erosion can result in leaching in nutrients from soils and increase riverine nutrient concentrations (Gabellone, Claps, Solari, & Neschuk, 2005). Any anthropogenically driven changes can result in augmented nutrient inputs, altered flow regime or a differing species composition and these all affect the time it takes for invertebrate communities to recover after a high flow event (Bertrand, Whiles, Gido, & Murdock, 2013). The physical changes that can be wrought on a river through the action of these more frequent floods of greater magnitude can result in altered invertebrate and fish communities. The effects of changes in a river's flood regime are important to consider for management and conservation.

Flood effects

Flooding, or periods of high flow are an integral part of the overall ecological health of a stream. Increased flow from rainfall, snowmelt and dam release can result in flooding. Floods are high flow events that overtop stream banks, whereas a 'spate' are periods of high flow that do not overtop (Russell Death, 2008). However, any alteration in flow discharge is likely to affect the invertebrate community so the term 'flood' is used here to refer to any high flow event which impacts the invertebrate community. During a flood, increased volume and velocity of the water means substrate may be subjected to increased shear forces which result in displacement. Additionally the water may also scour or abrade substrates, removing periphyton, macrophytes and bryophytes as well as insects clinging to rocks (Lake, 2000; Christoph D Matthaei, Arbuckle, & Townsend, 2000). Higher flows will also move large pieces of debris and organic matter such as leaves in streams with planted riparian margins. This results in a reduction of invertebrate biomass, density and diversity as insects are moved into the water column and washed away or are crushed by debris and mobile substrate (Robinson, Uehlinger, & Monaghan, 2003) This disturbance results in habitat changes within a stream. Streambed morphology may be altered and substrate material moved by the higher volume of water may cause infilling or scoured streambeds (Lake, 2000). While flooding may appear to have a deleterious effect on the faunal composition of rivers for a period after a high flow event, flooding is considered one of the most important driving forces of macroinvertebrate community composition (Lake, 2000; Resh et al., 1988).

New Zealand freshwater issues – nutrient enrichment

Nutrient enrichment as a result of anthropogenic activity is a growing issue worldwide. In New Zealand, nitrogen and phosphorus are the main contributors to nutrient enrichment in waterways. There are a number of ways that surplus nutrients can enter waterways: the excess application of fertiliser on both agricultural and urban land, the production of manure and urine by farm animals (Carpenter et al., 1998) and by other urban activities such as waste water treatment and construction (Novotny & Olem, 1994). These excess nutrients can then be carried off the land into waterways during rainfall, or phosphorus can leach through the soil into aquifers (Sims, Simard, & Joern, 1998).

Pollutants from urbanisation also contribute to nutrient enrichment in run-off (Carpenter et al., 1998). This type of pollution is attributed to diffuse sources such as lawn fertiliser and animal waste run-off as well as construction sites (Novotny & Olem, 1994). In a study by Chetelat, Pick, Morin, and Hamilton (1999) it was found that there was a strong positive correlation between algal biomass and phosphorus concentration. This indicates algal biomass may grow to eutrophic levels that impact river health in rivers that have a high enough phosphorus concentration.

Nitrogen and phosphorus effects on periphyton in waterways impact macroinvertebrates in multiple ways (Liess et al., 2009). If adequate light is present then plant growth will increase, and provided there is enough substrate and carbon available this will increase microbial activity and algal biomass (Pearson & Connolly, 2000). This will increase inputs to the detrital foodweb, enhance the condition of invertebrates, increase secondary productivity and these will result in changes in species diversity. (Pearson & Connolly, 2000). In aquatic systems, species richness is negatively correlated with increased productivity (Rosenzweig, 1995). This may lead to increases in invertebrate species evenness and richness (B. Biggs, Stevenson, & Lowe, 1998). However, if too many nutrients are added there is likely to be a shift towards a community dominated by species tolerant to eutrophication (B. J. Biggs, 2000). Bedford et al. (1999) noted that reduced diversity was common in streams with high levels of nutrient enrichment, and that these conditions favoured generalist species and may exclude more specialist species (Bedford, Walbridge, & Aldous, 1999). These conditions may increase primary production to such a level that it degrades stream health. Degradation effects may include a loss in aesthetic value (Wharfe, Taylor, & Montgomery, 1984), as well as the smothering of substrate by algae resulting in the loss of pollution sensitive invertebrate taxa (Quinn & Hickey, 1990). This increase in primary production can also degrade water quality, causing higher pH and lower dissolved oxygen, resulting in fish death (Quinn & Gilliland, 1989).

Why are benthic macroinvertebrates important for aquatic health?

Within freshwater ecosystems, benthic macroinvertebrates play a crucial role in nutrient cycling by determining how and at what rate organic matter is processed. This is the foundation of freshwater ecosystems. Each species is equipped to deal with variable conditions and different species are of different relative importance to these processes (Covich, Palmer, & Crowl, 1999). These species usually fall into four categories: primary producers, herbivores, predators or detritivores. Benthic invertebrates are estimated to process between 20% and 73% of riparian leaf litter inputs in a river's headwaters (Wallace & Webster, 1996). These species transform organic detritus into nutrients which can be used by primary producers. These benthic invertebrates release nutrients into solution by their excretion, feeding activities and burrowing. Other organisms such as bacteria, algae, fungi and macrophytes take up these now useable nutrients and growth and production is accelerated. The increased growth of these organisms in turn provides food for the benthic herbivores and omnivores, which are then consumed by predatory fish as well as providing a link to terrestrial ecosystems by also being consumed by terrestrial vertebrate consumers such as birds and reptiles. When the macroinvertebrate community composition changes because of pollution or habitat destruction or degradation, the trophic structure can be altered in such a way that the health of the waterway is impacted negatively (Sponseller, Benfield, & Valett, 2001).

Resilience

Resilience in an ecological context is defined as the time taken for an ecosystem to return to equilibrium following a disturbance event (Folke et al., 2004). In freshwater ecology the term resilience is usually used to refer to the aquatic macroinvertebrate community's ability to recover from a disturbance such as flooding (Peterson, Allen, & Holling, 1998). A central theme of freshwater ecology is that a rivers flow regime is fundamental to the structure of the macroinvertebrate community. The magnitude, duration and frequency of floods affects the resilience of invertebrates (Lake, 2000). A study by Suren and Jowett (2006) suggested that a larger magnitude flood would alter the invertebrate community composition more than a flood of smaller magnitude, but that the taxon richness would be less affected than invertebrate relative density. The magnitude of a flood will affect the time it takes for an invertebrate community to recover, the general trend being that the larger the flood, the longer the invertebrate community will take to recover. Flecker and Feifarek (1994) found a statistically significant positive relationship between invertebrate density and days since the last high flow event.

The frequency of high flow events also affects the resilience of the invertebrate assemblage. Assemblages that exist in more frequently disturbed streams recover more quickly from these events than assemblages in an infrequently disturbed stream. This is likely to be because more frequently disturbed sites are dominated by species who have the appropriate adaptations to persist in fluctuating conditions (Russell Death, 1996). These adaptations to frequent disturbance are small body size, generalist habits, dorsoventrally flattened bodies, two or more life stages outside the stream and corresponding high adult mobility (C. Townsend, Doledec, & Scarsbrook, 1997). It has been theorised that river systems with more frequent and larger floods would have a community assemblage and populations that demonstrated greater resilience and resistance to future disturbances than those adapted to more stable environments (Connell, 1978). Invertebrates that inhabit streams and rivers have a number of evolutionary adaptations that allow them to be resilient enough to successfully complete their lifecycles.

Resistance

Resistance is defined as the ability of stream organisms to resist the effects of a disturbance event. One of the most important ways invertebrates may resist disturbance is by using refugia. Refugia are places or parts of an organism’s life cycle that reduce disturbance related mortality (Lancaster & Belyea, 1997). Organisms may actively move or be moved by flow into areas that are less affected by the disturbance such as holes, immovable obstacles, interstices or debris that offers protection (Lake, 2000). Lifecycle refugia may be successful for species in a stream. Zwick (1996) proposed that certain species of stonefly use egg dormancy to create a “seed bank” within a stream. These eggs could provide the means to recolonise after a disturbance event. Certain organisms as well as first instar larvae may take refuge in the hyporheic zone and recolonise the surface once the disturbance is over. However, whether invertebrates seek refugia in the hyporheic zone is still contentious and the very few studies that have been carried out on the topic have found contrasting results. Some studies have noted the movement of certain macroinvertebrates into the substratum to avoid scouring suggesting that invertebrates do seek refuge in the hyporheic zone (Dudley Williams & Hynes, 1974; Hauer & Stanford, 1982; Poole & Stewart, 1976). Studies conducted in the Rhone River found that while some areas of the hyporheic zone provided refuge for invertebrates, differences in habitat complexity, sediment and geomorphology of the river ultimately decided riverine suitability as refugia (Dole-Olivier, 2011; Dole-Olivier, Marmonier, & Beffy, 1997). However, other investigations by R. G. Death et al. (2015) and Olsen and Townsend (2005) found species may be able to resist being swept away during a flood. Anthropogenic activities which result in streams with increased nutrient inputs, altered flow regime or different species composition impact the time it takes for invertebrates to recover after a high flow event (Bertrand, 2013).

Understanding the relationships between nutrient loading, disturbance regimes and macroinvertebrate community composition is important because an ecosystem must have all components functioning correctly for the stream to remain healthy. It is also important for preventing the loss of species not only that rely on streams for habitat but also terrestrial species that rely on streams for sustenance or completion of lifecycles. From an anthropological perspective, increasing human population growth dictates that the need for freshwater also increases. In the past, freshwater systems have been largely neglected and the resulting damage and alterations of waterways has been dramatic. The need to understand the relationship between macroinvertebrate invertebrate community structure and nutrient enrichment has never been more urgent and will allow us to construct a more complete picture of the riverine ecosystem. This will provide us with the means to develop tools for mitigating and preventing further damage and to inform policy and decision making. Understanding how nutrient loading in a stream affects macroinvertebrate community structure may allow for the adoption of optimum management practises across the different water use types to maintain healthy ecosystems. This study aims to determine what the most important driver of benthic invertebrate community structure is in four Taranaki streams. To determine this, the invertebrate community in each stream will be sampled to determine its composition, and physicochemical data will be used to gain an understanding of the disturbance and nutrients regimes in the streams.

Chapter 2: The effects of nutrient enrichment and disturbance regime on stream macroinvertebrate communities

Introduction

Ecological effects of nutrient loading in waterways are relatively well explored in New Zealand (Anderson, Glibert, & Burkholder, 2002; Lemly, 1982; McDowell, Larned, & Houlbrooke, 2009). In New Zealand, agriculture is a major contributor of non-point source nutrient addition to streams (Davies-Colley, Nagels, Smith, Young, & Phillips, 2004) with nitrogen and phosphorus being the major contributors to enrichment. Increasing levels of nutrients generally lead to an increased level of primary productivity. If adequate light is present then plant growth will increase and provided there is sufficient substrate and carbon available this will increase microbial activity and algal biomass (Pearson & Connolly, 2000). This algal and microbial biomass forms the basis of food webs in streams along with the input of detritus. In streams, increased nutrient levels can have a positive effect on primary production at the base of the food web (Cross, Wallace, Rosemond, & Eggert, 2006). This in turn enhances the productivity of intermediate and top consumers within that food web. Cross et al. (2006) found in an experiment examining the relationship between productivity and nutrient enrichment that invertebrate productivity increased with nutrient enrichment. Abundance, biomass and density of invertebrates also increased with increased nutrients. However, the addition of excess nutrients can cause overgrowth of algae or macrophytes (Anderson et al., 2002). Eutrophication can also cause fluctuations in oxygen that can be harmful or lethal to aquatic organisms as well as reducing suitable aquatic habitat for many species. Long term eutrophication can alter the macroinvertebrate community composition as pollution sensitive taxa are lost (Quinn & Hickey, 1990). Bedford et al. (1999) noted that reduced diversity of invertebrates was common in streams with high levels of nutrient enrichment because these conditions favoured generalist species and excluded specialists.

Floods (also referred to as disturbance) are arguably the most important formative factor for streams' channel and floodplain structure and their associated macroinvertebrate and riparian communities (Milner, Robertson, McDermott, Klaar, & Brown, 2013). In many regions of the world climate change is altering flood regimes: in New Zealand more rainfall is expected to increase flood magnitude and frequency (Sansom & Renwick, 2007). Flooding can be defined in an ecological context as a high flow event which results in the overtopping of stream banks, and the term spate is reserved for periods of high flow which do not result in overtopping but may have effects on the stream community nevertheless (Russell Death, 2008). During a flood, increased volume and velocity of water means substrate may be subjected to increased shear forces which can result in displacement of the substrate. Additionally the water may also have a scouring or abrasive effect, removing periphyton, macrophytes and bryophytes, large pieces of debris and other organic matter as well as displacing macroinvertebrates clinging to the substrate (Lake, 2000; Christoph D Matthaei et al., 2000). These effects result in a decrease in invertebrate biomass, density and diversity as insects are moved into the water column and washed away or are crushed by debris and mobile substrate (Robinson et al., 2003). But an important characteristic shared by streams worldwide is the rapidity of recovery following a flood event (Lake, 2000; Resh et al., 1988; Yount & Niemi, 1990). Small scale experiments conducted in streams found that recovery of macroinvertebrate communities can take from 4 days up to 6 weeks (RG Death, 1996; C. D. Matthaei, Uehlinger, Meyer, & Frutiger, 1996; Yount & Niemi, 1990) but this can be affected by many factors. The principal methods of recolonization of macroinvertebrates are drift from upstream, walking along the substrate and aerial recolonization (C. D. Matthaei et al., 1996; Reice, Wissmar, & Naiman, 1990; Resh et al., 1988). The magnitude and frequency of high flow disturbance events influences the macroinvertebrate community composition (Boulton, Peterson, Grimm, & Fisher, 1992).

Multiple environmental stressors simultaneously affect many freshwater ecosystems. While we may have an understanding of individual factors that influence stream invertebrates, the combined effects of these stressors are often unpredictable (Christoph D Matthaei, Piggott, & Townsend, 2010). Disturbance and nutrient enrichment are two important factors affecting the structure of macroinvertebrate communities. Scarsbrook and Townsend (1993) predicted that habitats experiencing frequent disturbance should be characterised by low invertebrate species diversity and a low biomass of periphyton whereas habitats with a lower frequency of disturbance should have a diverse community of sedentary and more specialised species as well as a higher biomass of periphyton. Frequency and magnitude of disturbance can also influence the benthic invertebrate community. Streams experiencing more frequent disturbances may contain communities that are dominated by highly mobile and generalist species such as the leptophlebiid mayfly *Deleatidium*, as well as *Austrosimulium* (Diptera, Simuliidae), and members of the Chironomidae and Oligochaeta. In a study of 97 midwestern United States streams, Riseng, Wiley, and Stevenson (2004) found that the relationship between nutrient concentrations, primary producer biomass and algal biomass were mediated by the hydrological disturbance regime. The response of the benthic invertebrate communities depended on an interplay of these factors.

Following flooding, nutrient enriched streams may recover faster than those with lower nutrient levels. Weng, Mookerji, and Mazumder (2001) found that the periphyton and invertebrate communities in experimentally enriched sections of stream recovered more quickly after a period of high flow. However, a study by Demchick (2014) found that streams with a lower level of nutrient enrichment recovered more quickly following flooding than the streams with higher nutrient levels. A possible explanation for this is that areas which have a higher level of nutrient loading experience faster growth of the primary producers, leading to a more readily available food source for the higher trophic levels. Streams with abundant periphyton biomass did not have higher growth rates so took longer to recover to pre-flood levels than streams that were accustomed to lower periphyton biomass. This illustrates the linkage between primary production, disturbance patterns and the macroinvertebrate community composition. Long term changes in stream nutrient status can affect the invertebrate community structure. In an experiment by Gafner and Robinson (2007) it was found that invertebrate community structure was altered in the stream sections with nutrient enrichment. These enriched sections experienced a shift to a community assemblage dominated by chironomids.

There has been a growing awareness of the decline in water quality in New Zealand and with this, growing concern about the associated biodiversity loss and declining ecological health (Jacobson et al., 2014). Over the last 25 years, there has been a decline in water quality at many monitored sites across the country (Ballantine & Davies-Colley, 2014; Environment, 2013; Verburg, Hamill, Unwin, & Abell, 2010). As the agricultural industry in New Zealand has intensified, water quality has become most degraded at lowland sites where catchments are dominated in landuse by agriculture (Ballantine & Davies-Colley, 2014; Foote, Joy, & Death, 2015; Harding, Young, Hayes, Shearer, & Stark, 1999; Larned, Scarsbrook, Snelder, Norton, & Biggs, 2004; Nagels, Davies-Colley, Donnison, & Muirhead, 2002). Nationwide, median concentrations of *E. coli* exceed guidelines for both ecological health and human health at many monitored sites, and 60% of sites have increasing levels of dissolved reactive phosphorus and dissolved inorganic nitrogen (Larned et al., 2004). New Zealand lakes are similarly degraded

– 32% of lakes are now classified as polluted by nutrients and 84% of lakes in pastoral areas are the same (Verburg et al., 2010). With surface and groundwater inextricably linked it is inevitable that their health has also suffered. Thirty-nine percent of monitored sites have rising nitrate levels and 21% of these sites have pathogen levels that exceed the safe levels for human drinking water (Daughney & Wall, 2007).

Benthic invertebrates are commonly used as an indicator of ecological condition and water quality (J. Stark, Boothroyd, Harding, Maxted, & Scarsbrook, 2001). They are an ideal indicator of water quality as they are found in almost every freshwater environment and different taxa have differing tolerances to pollution and other ecological impacts (J. Stark et al., 2001). It is also inexpensive and simple to collect and identify invertebrates relative to other available methods. Water quality monitoring is fast becoming an essential part of freshwater science, owing to the rapid expansion of dairying and urbanisation over the past few decades. With this has come declines in waterway health and a growing awareness by the public of these facts. The public, non-governmental and governmental agencies alike realise the seriousness of the situation and are lobbying for the improvement in the health of New Zealand's waterways. To prevent and mitigate future damage requires a firm grasp on the current situation and the factors that do and will influence the ecological health of streams. There are often many factors that work in concert to determine stream health as discussed earlier and it can be difficult to disentangle one effect from another or predict what the combined effect of several impacts may be. This study aims to go a small way towards investigating the interplay between a rivers disturbance regime, nutrient inputs and the impacts this has on the macroinvertebrate community. The structure of the invertebrate community of a stream is a stable indicator of the waterways health. This is important for practical management and can be used to inform policy decision making as well as resource allocation from regional councils.

The aim of this study is to investigate how the relationships between flow and nutrient enrichment in four Taranaki streams affect the macroinvertebrate community composition.

Study questions

1. How does flow regime and nutrient enrichment affect the macroinvertebrate community composition?
2. Which factor is the most important determinant of macroinvertebrate community composition?

Methods

Study sites

Over 300 streams run from the flanks of Mt Taranaki in a near perfect radial pattern. With over 500 named streams, the rivers in this region have short tributaries contained by valleys. This means that in general these rivers carry a high sediment load from surrounding hill erosion. Egmont National Park supplies these streams and rivers with a near constant flow of water even in drier periods. These rivers are used extensively for agriculture, industry and community water supplies as well as for leisure activities. The extremely fertile volcanic ash soils of the ring plain lend themselves to intensive pastoral farming, particularly dairying. This abundance of water combined with extremely fertile soils means that dairy farming is most intensive on the flat plains and thus this area lends itself to a study investigating the effects of nutrient enrichment and altered physical habitat.

Four streams in the Taranaki region were included in this study (Fig. 1). Three sites on each stream were sampled: one close to the National Park, one close to the coast and one in between. The Waingongoro River, Kaupokonui River, Inaha Stream and Taungatara Stream were sampled between October and November 2017. Photos of some of the sites are presented in Fig 2 – Fig 5.

Figure 1: Map of the 12 sample sites on the four rivers, Taungatara Stream (TG), Kaupokonui Stream (KP), Inaha Stream (Inaha) and Waingongoro River (WG)

Figure 2: A) Taungatara Stream upstream site (TG1), B) Taungatara Stream middle site (TG2)

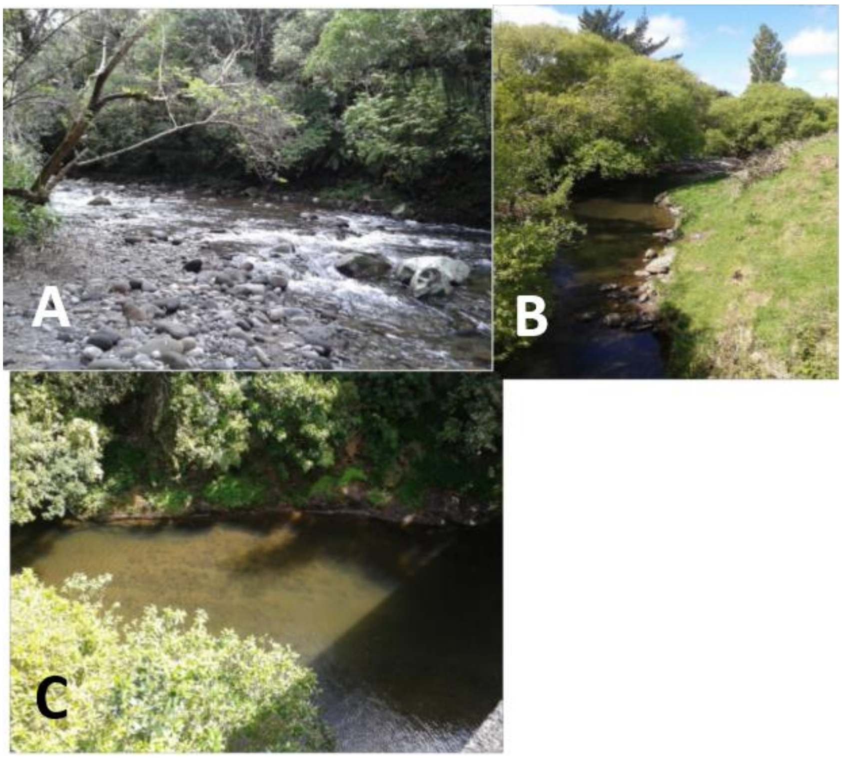

Figure 3: A) Waingongoro River, upstream site (WG1), B) Waingoro River middle site (WG2), C) Waingongoro River, lower site (WG3).

Figure 4: A) Inaha Stream, middle site (Inaha 2); B) Inaha Stream, lower site (Inaha 3)

Figure 5: A) Kaupokonui Stream upper site (KP1), B) Kaupokonui Stream middle site (KP2).

Sampling methods

Physicochemical characteristics

Physical and chemical characteristics of each stream were measured concurrent with macroinvertebrate sampling between October and November 2017. This included, depth, width and velocity measured with a Valeport Electromagnetic Flow Meter Model 801. These measures were taken at five equidistant points along each study reach. Water temperature and conductivity were measured using an Oakton meter (EuTech) with conductivity adjusted to 25°C. Streambed stability was assessed using the bottom component of the Pfankuch Stability Index (RG Death, 1995; Pfankuch, 1975). This index works by summing the score of five individual characteristics (scored according to their perceived importance) within pre-determined criteria.

The percentage of riparian area covered by the different vegetation types (scrub, crop/pasture, native, exotic forest, tussock and other) was estimated visually at each site. A single substrate index was derived from estimating percentage of each substrate size class (bedrock, boulders (>26 cm), cobbles (13 – 26 cm), pebbles (6 – 12 cm), gravel (0.2 – 6 cm), sand (< 0.2 cm), and silt/sand) and summing the midpoint value of the size classes weighted by their proportional cover. Bedrock was assigned a nominal value of 400mm for calculation purposes.

The percentage of macro-habitat types (pool, run, riffle) were visually assessed along the study site and substrate embeddedness was ranked on a scale from 1-4 (1 being loosely packed and 4 being a hard-packed clay/silt). The condition of the riparian margins was assessed using the Waterway Self-Assessment Form (Polglase, 2000). The Waterway Self-Assessment Form was developed as a tool for landowners to assess the riparian areas of their own land, and is a useful tool for pinpointing problem areas of waterways and monitoring progress of management programs.

Macroinvertebrate sampling

At each site five randomly placed Surber samples (area of $0.01m^2$, 250µm mesh) were collected in riffles. The samples were preserved in 70-80% isopropanol for transport back to the laboratory. For processing, samples were rinsed through a 500µm mesh sieve, and retained macroinvertebrates were identified and counted. Macroinvertebrates were identified to the lowest possible taxonomic resolution, except for Chironomidae which were identified only to tribe.

Periphyton sampling

Five random stones were collected next to each of the Surber samples. Stones were kept in river water on ice in the field. At the laboratory, the water was removed and the stones were frozen at -20°C. Pigment was extracted from the stones by covering them with 90% acetone and keeping at 4°C for 24 hours in the dark. As a proxy for chlorophyll α levels, absorbency was measured using a Cary 50 Conc UV-Visible Photospectrometer and chlorophyll α was calculated using the method outlined in Steinman, Lamberti, Leavitt, and Uzarski (2017).

Biotic indices

Biological indices are a tool to assess the relationship between the macroinvertebrate community and water quality. There are four standard indices which were used here: the Macroinvertebrate Community Index (John Douglas Stark, 1985; John D Stark, 1993)

- Macroinvertebrate Community Index (MCI) (John Douglas Stark, 1985) is an index used to measure waterway health in New Zealand. It uses the presence or absence of stream macroinvertebrates as a biological indictor of stream health. The index assigns a number to each taxon according to its sensitivity to organic pollution (1 being least sensitive, 10 being most sensitive) and these are used to calculate a score. Scores of above 125 are considered 'pristine', while scores of 75 or less are considered 'severely polluted' (Table 1). This index was originally developed for use in Taranaki hard-bottomed streams so is particularly relevant to this study.
- Quantitative Macroinvertebrate Community Index (QMCI) is similar to the MCI but it takes into account the number of individuals of each species collected. Rather than using the presence or absence of taxa like MCI does, QMCI uses densities and so is not sensitive to finding taxa that are represented by only one or two animals. Given that the

densities of some taxa may reach tens of thousands per square meter of stream bed, QMCI is usually given more credence when assessing any potential effects (Wright-Stow & Winterbourn, 2003)

Table 1: Interpretations of scores for MCI, QMCI, %EPT taxa and % EPT individuals. EPT values are not established thresholds and are instead based on R Death pers comm.

Interpretation	MCI	QMCI	%EPT taxa	%EPT individuals
Clean water	>120	>6.0	>60	>60
Mild pollution	100-120	5.0-6.0	10-60	10-60
Moderate pollution	90-100	4.0-5.0		
Severe pollution	<80	<4	<10	<10

%EPT taxa: Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddis flies) are three groups of insects that are generally sensitive to pollution. The %EPT is the proportion of taxa collected at a site that belong to one of these three orders. A score of 100% would indicate that every individual collected in that sample were mayflies, stoneflies or caddis flies and thus that the stream is likely to be very healthy. Conversely, 0% would indicate that no individuals in the sample belonged to one of these groups and that stream health is likely to be very poor. Two genera of caddis fly, *Oxyethira* and *Paroxyethira* are generally excluded from this index as they are generally quite tolerant to pollution. Values of between 60 and 10 (corresponding to good and bad) indicate mild to moderate pollution.

- %EPT individuals: This index is similar to %EPT taxa but instead measures the proportion of individuals collected that are mayflies, stoneflies and caddis flies. *Paroxyethira* and *Oxyethira* are also excluded from this index. The same scoring system as %EPT is applied for this index.

Data analysis

Differences in community composition between the four waterways (Waingongoro, Kaupokonui, Inaha and Taungatara) were examined by conducting an analysis of similarities (ANOSIM) using the ANOSIM routine in the PRIMER statistical package following log (x+1) transformation of the data (Clarke & Warwick, 2001). ANOSIM is a non-parametric method that assesses whether the average similarities between samples within a group are closer than the average similarities of all pairs of replicates within all groups (Clarke & Warwick, 2001). The mean density of taxa that contributed most (>50%) to the dissimilarity between the three groups was analysed using the SIMPER procedure in the PRIMER statistical package.

To complement the habitat data collected in the field, GIS information on nutrients, flow regime, catchment geology and topography, temperature, shading and deposited sediment were all examined. Catchment geology, topography and temperature were obtained from the FENZ database and modelled sediment data from Clapcott, Goodwin, and Snelder (2013). This data was incorporated and used for the ANOSIM analysis as outlined above.

Flow data was obtained from Taranaki Regional Council's website and GIS data was also obtained to determine the mean annual flood (Clausen & BIGGS, 1997; Griffiths & McKerchar, 2012). This method uses several river metrics including length, slope and Manning coefficient of roughness in the main channel and provides an estimate of the mean annual flood size in cubic metres per second (m^3/s). Measures from each site on each stream averaged to obtain a single flood regime value for each stream. Using this method allows the relationship between the disturbance regime and how it relates to nutrients and periphyton to be examined.

Results

Physical characteristics

The sites included in this study ranged in width from 2.5 to 27.5 m, and depth varied from 25 to 37 cm. Conductivity ranged from 15 to 330 µS/cm. Four sites were dominated by small cobbles and gravel and seven were dominated by a mixture of large boulders with large and small cobble substrate. Two of the Inaha sites comprised mostly sand and silt. Pfankuch scores ranged from 20 to 60. Table 2 presents all the physical characteristics of the sites.

Table 2: Summary of physical data for all streams sampled October-November 2017

Site	WG 1	WG 2	WG 3	Inaha 1	Inaha 2	Inaha 3	KP 1	KP2	KP3	TG 1	TG 2	TG 3
Date	21/1017	3/11/17	27/10/17	2/11/17	2/11/17	27/10/17	2/11/17	3/11/17	27/10/17	2/11/17	2/11/17	27/10/17
Easting	6204324	1710611	6180332	1703676	1822318	6182160	1698088	1710606	6190719	1684494	1681878	6191248
Northing	2615847	5634800	2613994	5635423	5525926	2610450	5639222	5634792	2607324	5638398	5634429	2586784
Conductivity mS m^{-1} at 25 °C	90	15	230	180	230	330	90	100	150	120	140	180
Temperature °C	11.4	14.2	14.3	15.8	14.2	14.3	14.1	14	15.9	15.9	17.2	15.5
Width (m) (avg)	5.5	17	27.5	2.5	4	5	15	15	20	4	25	20
Depth (cm)(avg)	24.4	31.8	30.4	33	27.4	29.4	25.2	36.8	33.8	23.4	23	33.6
Velocity (avg)(m/s)	0.512	0.304	0.39	0.234	0.612	0.410	0.646	0.287	0.598	0.872	0.72	0.552
% Pool	30											
% Riffle	40	5	50		10		95	5	100	100	50	90
% Run	30	95	50	100	90	100		95			50	
% Rapid							5					10
% Sand & Silt (<8mm)	0	35	0	75	0	100	0	0	2	0	0	1
% Gravel (8-16mm)	0	1	4	1	40	0	0	0		0	0	3
% Pebbles (17-64mm)	29	44	22	13	49	0	8	24	16	8	54	17
% Small Cobbles (65-128mm)	40	18	26	8	7	0	23	47	21	14	39	17
% Large Cobbles (129-300mm)	19	2	4	2	3	0	50	9	8	15	7	6
% Boulders (>300mm)	12	0	44	1	1	0	19	20	53	63		56
Pfankuch	28	35	26	50	36	60	36	29	20	26	32	26

Habitat quality measures

Figure 6 presents the Waterway Self-Assessment Score for each site at the four streams. The Waingongoro River had the highest Waterway Assessment score. The Kaupokonui and Taungatara Streams interestingly had higher scores in the lower reaches compared to the upper reaches. Inaha Stream Site 3 had the lowest score, mainly due to the smaller riparian zone compared to the other waterways. Figure 7 shows the relationship between average MCI scores and Waterway Self-Assessment Scores, however this was not statistically significant (R=0.20, P=0.143).

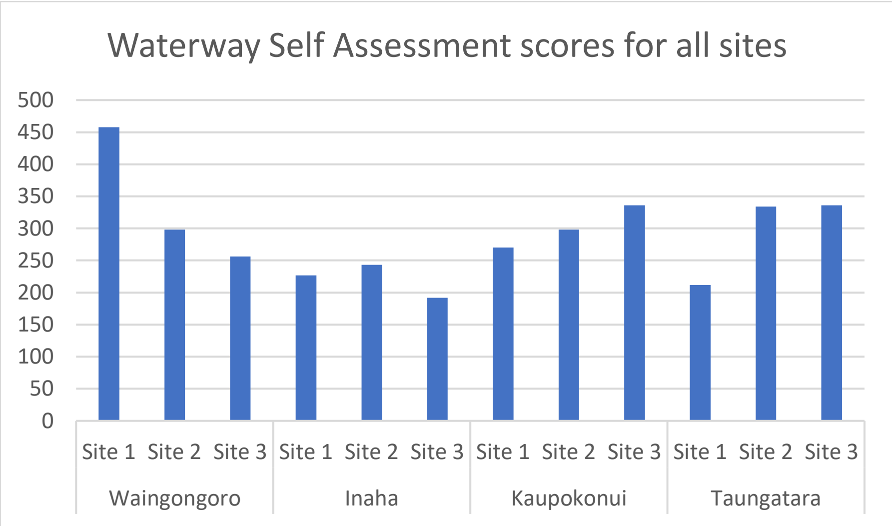

Figure 6: Waterway Self-Assessment scores for all three sites at four Taranaki waterways. Scores <200 indicate poor riparian condition, 200 is good condition and >390 indicates excellent riparian condition.

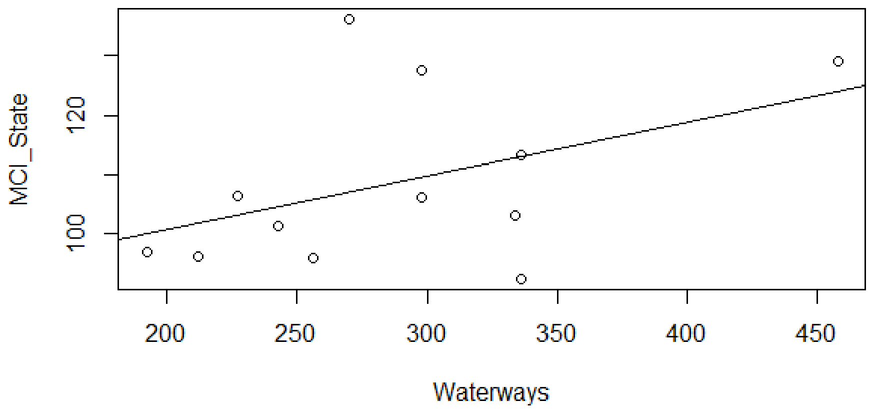

Figure 7: Average MCI for 5 replicate Surber samples collected at 12 Taranaki Stream sites in November 2017 plotted against Waterway Self-Assessment scores.

Substrate – sorting, skew and Kurtosis

In addition to the Waterway Self-Assessment conducted above, pebble counts of the sediment were used to determine the particle size distribution within the four waterways at each of the three sites. There is a general trend of scoring greater than 1 in the lower reaches (Site 2 and 3). This indicates that as waterways become larger, the substrate becomes more poorly sorted while the upper reaches were more moderately sorted. The large negative values found for the skewness measure indicate that there was a larger proportion of coarse material (i.e more large cobbles than fine gravel). Conversely, positive skew values indicate the presence of finer particles (fine gravel or sand) which was characteristic of Inaha Stream. When looking at the Kurtosis measure, all sites except Inaha Stream Site 2 showed leptokurtic sorting (better central sorting) while Inaha Site 2 showed platykurtic sorting where the sorting is better at the tail ends of the distribution.

Table 3: Summary of sediment data for Waingongoro River (WG), Inaha Stream (Inaha), Kaupokonui Stream (KP) and Taungatara Stream (TG).

Reach	D_{50}	D_{84}	Sorting	Skewness	Kurtosis
WG Site 1	92	188	0.95	0.83	1.74
WG Site 2	35.2	72.53			
WG Site 3	120	322.99	1.22	-0.96	3.51
Inaha Site 1		46			
Inaha Site 2	18	40	0.89	4.24	0.90
Inaha Site 3	N/A	N/A	N/A	N/A	N/A
KP Site 1	192.6	272.74	0.80	-1.90	1.20
KP Site 2	90.84	277.2	1.05	2.07	1.68
KP Site 3	262.23	330.07	1.09	-5.86	2.12
TG Site 1	277.87	335.08	0.85	-4.07	1.16
TG Site 2	61.87	104	0.79	0.89	1.05
TG Site 3	267.35	331.71	1.36	-10.49	3.54

Flood magnitude

Flow data obtained from Taranaki Regional Council and GIS layers to calculate the mean annual flood (measured in m^3/s) to rank the streams in order of disturbance frequency. Table 4 gives the mean annual flood values. The Waingongoro River had the highest mean annual flood size at 42.99m³/s and is the most disturbed stream. The Inaha Stream had the lowest mean annual flood size of 8.05m³/s; the least disturbed stream. Kaupokonui ranks second with a mean annual flood of 14.58m³/s and Taungatara Stream third with a value of 8.94m³/s.

Conductivity

Conductivity increases between site 1 and 3 on all four waterways (Fig. 13). This increase reflects the increasing amount of agricultural contribution to nutrient enrichment such as runoff, seepage, direct input of faeces and/or urine the further downstream travelled. There is a significant relationship ($R=0.45$, $P=0.017$) between increasing conductivity with decreased distance to the coast and upstream sites tended to have lower conductivity than downstream sites (Figure 8).

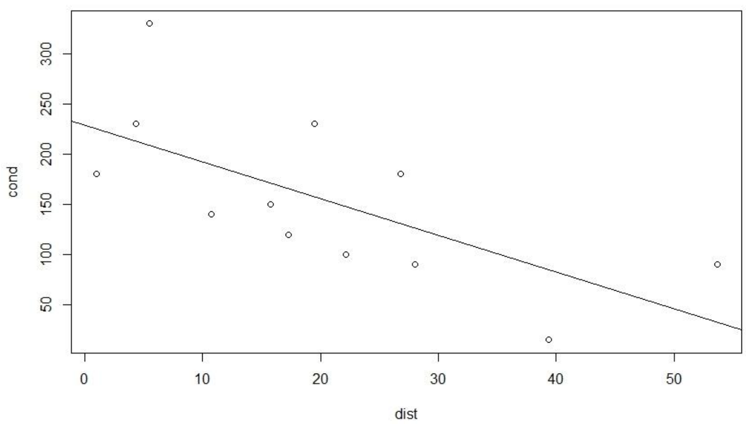

Figure 8: Graph showing relationship between conductivity (cond) and distance (dist) to the coast in four Taranaki streams sampled October-November 2017.

Periphyton

Chlorophyll α values for all sites are presented in Figure 9. Kaupokonui Stream showed an increase of periphyton from upstream to downstream. Taungatara Stream had the highest value at Site 2 but the values at Site 1 and 3 remained similar. The Waingongoro River showed the highest periphyton biomass in the midsection, while Site 1 (the upstream site) had the lowest periphyton biomass out of all sites sampled. Inaha Stream Site 3 was dominated by silt and gravel so periphyton sampling was not possible with the method used for this study. However Inaha Site 1 and 2 showed a slight decrease in biomass from upstream to downstream. Periphyton growth relies on nutrients being present in sufficient quantities to allow growth so it may be expected that there will be a relationship between the nutrients levels in a waterway and the amount of periphyton growth.

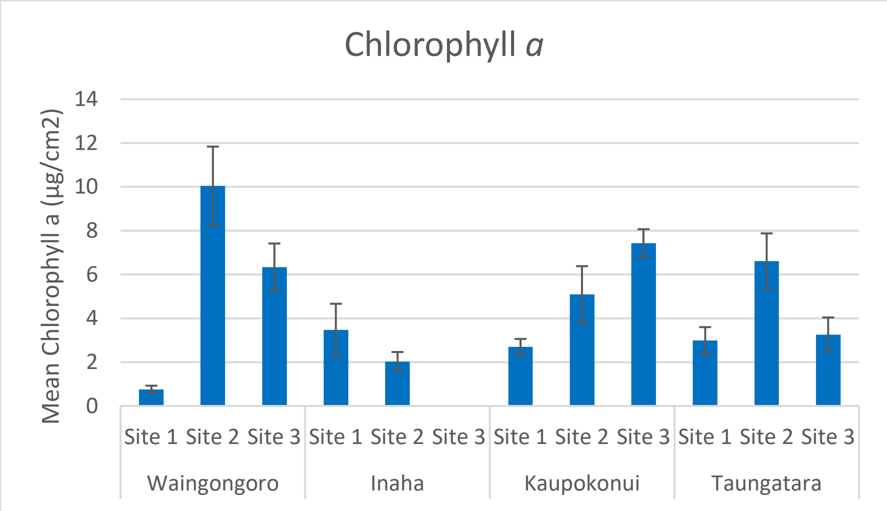

Figure 9: Mean chlorophyll α values for each site, measured as (µg/cm²) collected at 3 sites on the 4 Taranaki waterways.

Invertebrate community composition (diversity)

Sixty-nine taxa in total were collected from 12 sites. The macroinvertebrate community is dominated by Ephemeroptera (mayflies) and Trichoptera (caddisflies), with the exception of Inaha Site 3 and to a lesser extent, Waingongoro Site 3. The respective compositions by order for each site are shown in Figure 10.

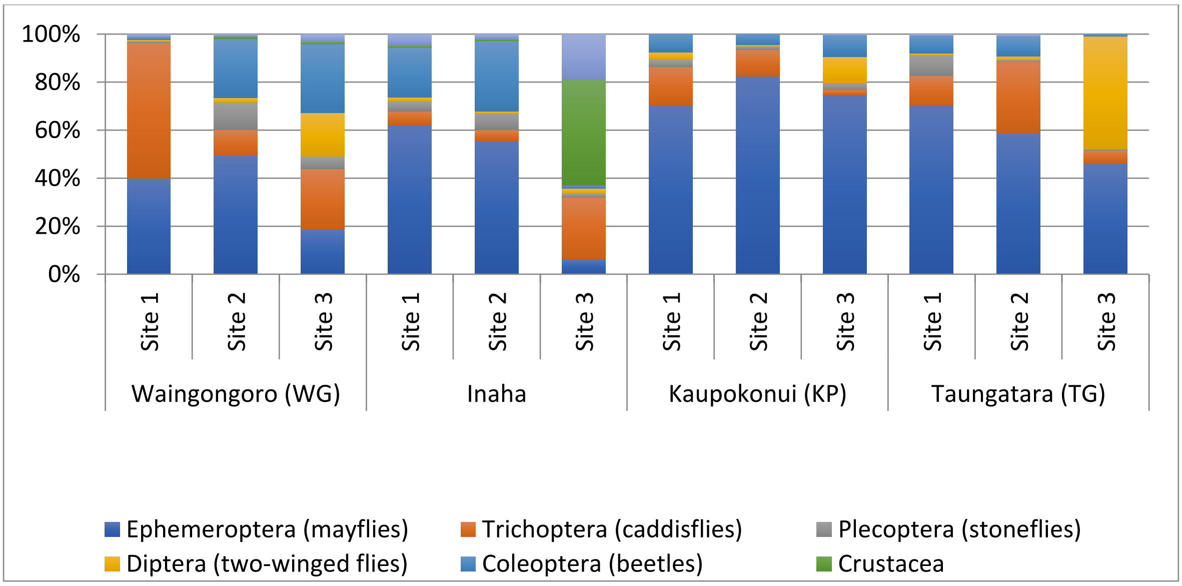

Figure 10: Relative abundance (%) of higher order taxa at three sites on four Taranaki waterways.

The most abundant taxa sampled were usually Ephemeroptera (mayflies) and Trichoptera (caddis flies) although the species collected differed. Coleoptera (beetles), Diptera (two winged flies) and Plecoptera (stoneflies) were also recorded at all sites but in lower numbers relative to mayflies and caddis flies and again the species collected differed by site. A range of other taxa were also collected: Crustacea were found at 5 sites and Oligochaeta at seven, as well as Odonata (damselflies and dragonflies) (1 site; Inaha Site 3), Hirudinea (leeches) (2 sites), Oligochaeta (worms) (7 sites), Megaloptera (dobsonflies) (6 sites), Mollusca (molluscs) (9 sites) and Acari (mites) at 2 sites. Kaupokonui Stream contained no species classified as 'other' at Site 1 or 2, only at Site 3 were these found.

The Waingongoro River was characterised by 37 species collected from the three sites. Predominant taxa included 'highly sensitive' members of Ephemeroptera (*Oniscigaster, Nesameletus*, *Deleatidium* and *Coloburiscus*), Plecoptera (*Zelandoperla*) and Trichoptera (*Helicopsyche, Beraeoptera* and *Orthopsyche*. 'Moderately sensitive' taxa included *Zephlebia* (Ephemeroptera), elmid larvae (Coleoptera), *Zelandobius* (Plecoptera), *Costachorema* and *Pycnocentrodes* (Trichoptera), *Archichauliodes* (Megaloptera), Amphipoda and Ostracoda (Crustacea) and *Aphrophila* (Diptera). Tolerant species included *Austrosimilium* (Chironomidae, Diptera), Orthocladiinae (Diptera) and *Hydropsyche-Aoteapsyche* (Trichoptera).

Taungatara Stream was characterised by 34 species. Predominant taxa included 'highly sensitive' taxa *Oniscigaster*, *Nesameletus* and *Deleatidium* (Ephemeroptera), *Zelandoperla* (Plecoptera) and *Beraeoptera*, *Olinga*, *Helicopsyche* and *Oeconesus* (Trichoptera); 'moderately sensitive' taxa *Zephlebia* (Ephemeroptera), *Archichauliodes* (Megaloptera), *Pycnocentrodes* (Trichoptera), *Diamesinae* (Chironomidae, Diptera), *Aphrophila* (Diptera); and 'tolerant' species *Aoteapsyche-Hydropsyche* (Trichoptera), *Chironominus* (Diptera) and Orthocladiinae (Diptera).

Kaupokonui Stream is characterised by 41 species. Predominant taxa included 'highly sensitive' taxa included *Oniscigaster, Nesameletus* and *Deleatidium* (Ephemeroptera). *Zelandoperla* (Plecoptera), *Beraeoptera* and *Helicopsyche* (Trichoptera); 'moderately sensitive' taxa *Zephlebia* (Ephemeroptera), Elmidae larvae (Coleoptera), Hydraenidae (Coleoptera), *Archichauliodes* (Megaloptera), *Pycnocentrodes* (Trichoptera), *Zelandobius*

(Plecoptera), *Aphrophila* (Diptera) and 'tolerant' species including *Hydropsyche-Aoteapsyche* (Trichoptera), Chironomidae and Orthocladiinae (Diptera).

Inaha Stream is characterised by 34 species. 'Highly sensitive' taxa collected included *Oniscigaster*, *Nesameletus* and *Deleatidium* (Ephemeroptera), *Beraeoptera* (Trichoptera) and *Zelandoperla* (Plecoptera). 'Moderately sensitive' taxa include *Archichauliodes* (Megaloptera), *Zephlebia* (Ephemeroptera), Elmidae larvae (Coleoptera), *Costachorema* and *Pycnocentrodes* (Trichoptera), *Zelandobius* (Plectopera), *Amphipoda* and *Ostracoda* (Crustacea) and *Aphrophila* (Diptera). 'Tolerant' taxa included *Austrosimulium* (Diptera), *Hydropsyche-Aoteapsyche* (Trichoptera), Oligochaeta, Orthocladiinae (Diptera) and *Potamopyrgus* (Mollusca).

EPT taxa

Waingongoro Site 1 had the greatest abundance of EPT taxa at 75.54% (Fig. 11) and comprising 98.7% of total individuals (Fig. 12) indicating good water quality. The lowest abundance of EPT taxa was at Inaha Site 3 with 34.83% with 25.45% of individuals at that site which suggests that the water in Inaha Stream is of moderate to low quality. This suggests that there is nutrient contamination in this waterway. Both %EPT taxa and %EPT individuals show a general trend towards lower percentages in lower reaches (ie Site 1 > Site 3). The most numerous taxa across all sites was the mayfly *Deleatidium* which was found at all 12 sites, *Beraeoptera* was also found at 12 sites and *Pycnocentrodes* found at 11 sites.

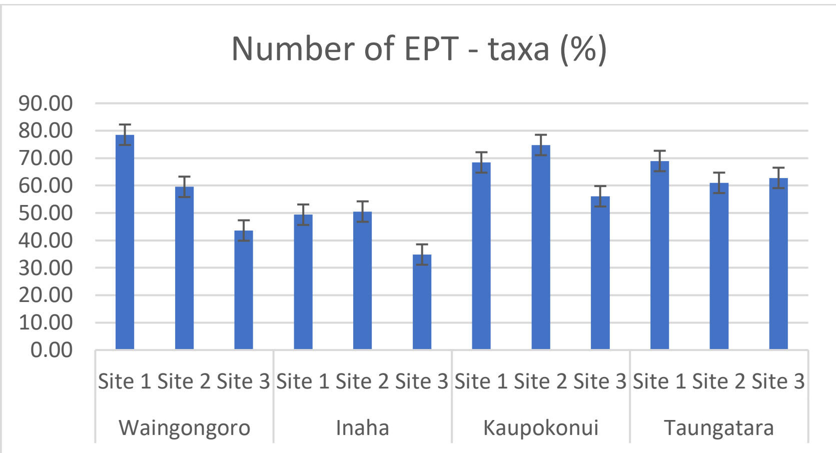

Figure 11: Mean percentage of EPT taxa found at three sites on the four Taranaki waterways

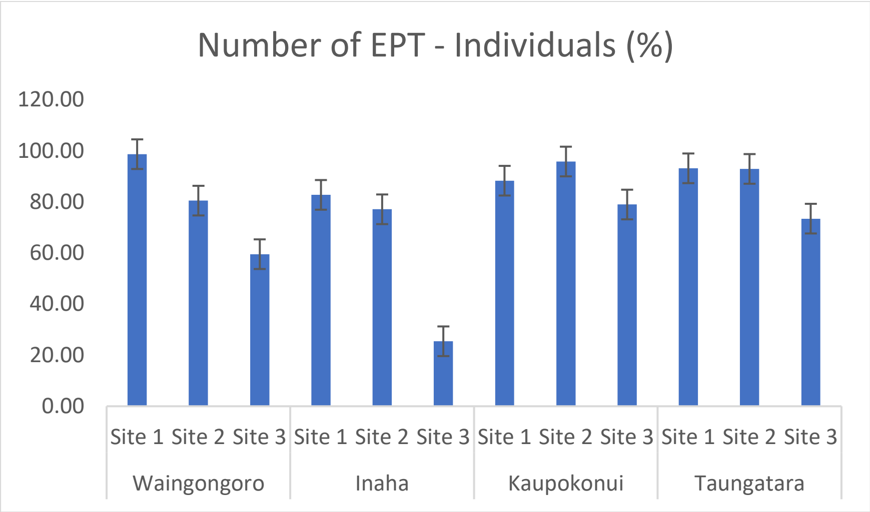

Figure 12: Mean number of EPT individuals found at three sites on four Taranaki waterways

Multivariate analysis of similarities

Changes in macroinvertebrate community composition along the length of each waterway can be seen in Figure 13 (Bray-Curtis distance plotted in an ordination). The closer together two samples are, the more similar their community composition. An analysis of similarities indicated the greatest differences between downstream sites (labelled 3), while there was no significant difference between the upstream sites (labelled 1 and 2) at each waterway.

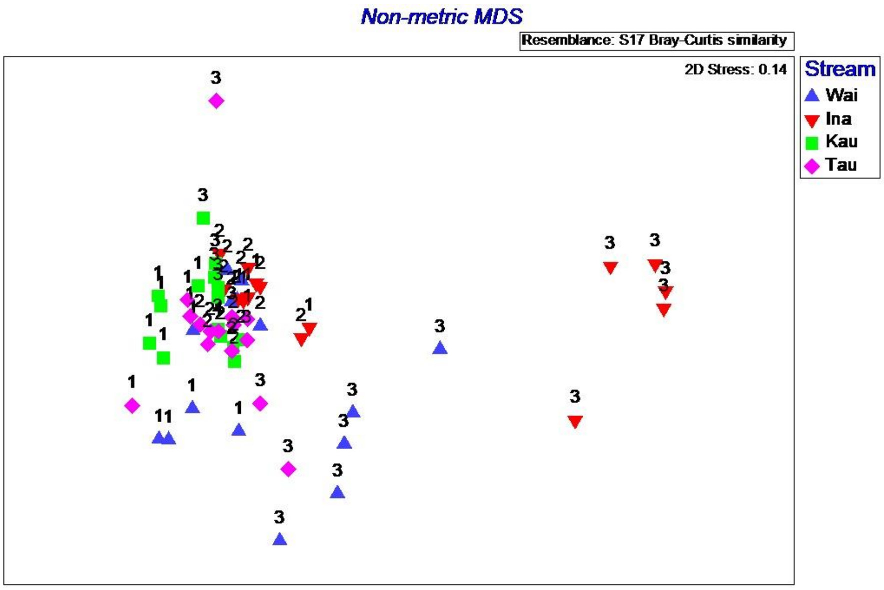

Figure 13: Multivariate analysis performed in PRIMER using Bray-Curtis similarity for each of the four Taranaki streams at three sites sampled in October-November 2017.

Water quality indices

MCI and QMCI values showed similar patterns across the 3 sites on the 4 waterways. Scores were generally higher in the upper reaches (Table 4, figure 14 & 15). Taungatara and Kaupokonui Streams had the highest scores of the four streams tested on average, and Inaha Stream the lowest, however there did not appear to be a significant difference between the four waterways.

Table 4: table of MCI and QMCI results by site

	QMCI			MCI		
Site	Site 1	Site 2	Site 3	Site 1	Site 2	Site 3
Waingongoro	Excellent	Excellent	Fair	Excellent	Good	Fair
Taungatara	Excellent	Excellent	Good	Excellent	Excellent	Good
Kaupokonui	Excellent	Excellent	Excellent	Excellent	Excellent	Excellent
Inaha	Excellent	Excellent	Fair	Good	Good	Fair

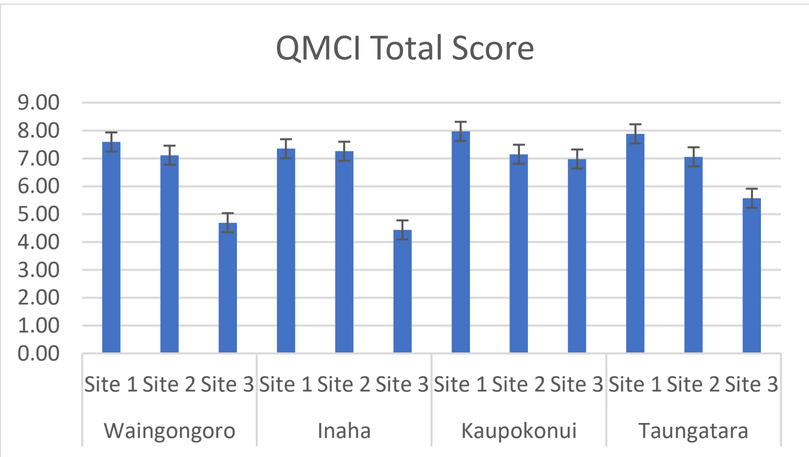

Figure 14: Mean QMCI scores for each of three sites at four Taranaki streams

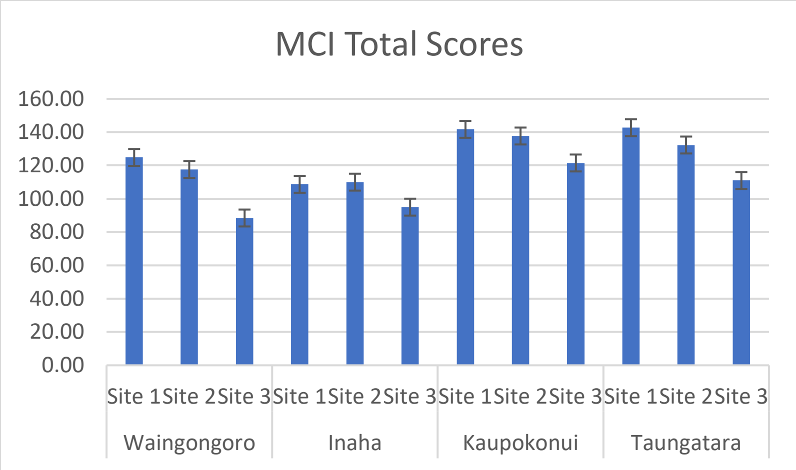

Figure 15: Mean MCI scores for three sites at the four Taranaki waterways

Nutrients

There is a strong relationship between the benthic macroinvertebrate community composition and the nutrient levels of a stream. In this study, dissolved reactive phosphorus (DRP) appeared to have the strongest relationship with MCI scores (Fig. 16). There is a downward trend in MCI scores with increasing DRP except for Inaha Stream where as DRP increased so did the MCI scores. Table 4 lists the values for DRP and nitrates by site.

Table 5: Dissolved reactive phosphorus (DRP) and nitrates (NO_3-N) for each site in mg/L-1.

	DRP	NO_3-N
Waingongoro 1	0.014841	0.157086
Waingongoro 2	0.021418	0.943983
Waingongoro 3	0.036098	1.586953
Inaha 1	0.021248	1.129167
Inaha 2	0.020903	1.511074
Inaha 3	0.018863	1.589467
Kaupokonui 1	0.013167	0.066296
Kaupokonui 2	0.015168	0.132567
Kaupokonui 3	0.018004	0.362467
Taungatara 1	0.022373	1.086621
Taungatara 2	0.021227	0.559279
Taungatara 3	0.022871	0.735606

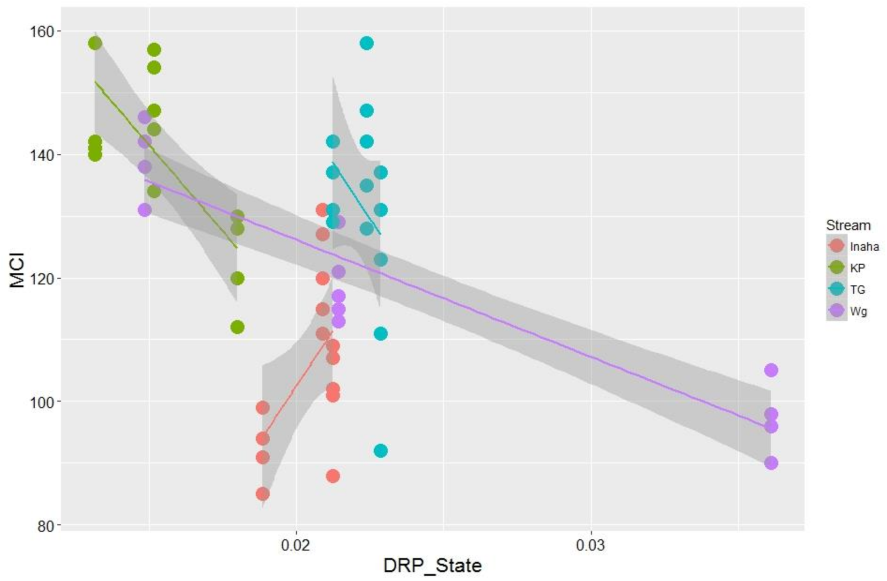

Figure 16: Average MCI collected at 12 sites in 4 Taranaki streams as a function of median dissolved reactive phosphorus (DRP).

Discussion

Generally, water quality declined between the upstream and downstream sites. Macroinvertebrate communities at sites in the mid and upper catchment sites comprised a greater proportion of taxa classified as sensitive (%EPT) to the effects of organic pollution than in the lower reaches of the catchments. These changes in macroinvertebrate community composition are likely driven by the effects of nutrient enrichment, increasing temperatures and increased periphyton biomass resulting in poorer water quality in downstream reaches. Inaha Stream appeared to be the most affected by agricultural activity perhaps because it is a smaller stream which originates from within the agricultural landscape as opposed to flowing off Mt Taranaki as the other stream in this study do. Inaha Stream also has a lower rate of disturbance relative to the other streams. Faunal composition generally changed between the upstream and downstream sites. Downstream macroinvertebrate communities tended to be more likely to be composed of species which are less sensitive to the effects of organic pollution such as chironomids and snails. Conductivity increased from upstream to downstream, indicating increasing nutrient enrichment downstream. Upstream sites were more similar to each other with communities dominated by more sensitive taxa including Ephemeroptera (mayflies) and Trichoptera (caddisflies) indicating higher water quality at the upstream sites less affected by human activity and resulting nutrient enrichment. This suggests water quality changes downstream due to nutrient enrichment affect the invertebrate community composition, however the Waingongoro River experienced no significant change in community composition between the upstream and downstream sites whereas the Inaha Stream experienced the greatest dissimilarity between the upstream and downstream sites. There was a strong relationship between dissolved reactive phosphorus (DRP) and MCI scores, with increasing DRP correlated with decreasing MCI scores except for the Inaha Stream where as DRP increased so did MCI scores. This may be due to the relative lack of disturbance in this stream which allows more algal growth to support more invertebrates for longer periods of time.

Flood frequency is a major driver of macroinvertebrate community composition and it is well known that invertebrate community composition differs significantly between stable and flood-prone streams (Barquín & Death, 2004, 2006; RG Death, 1995; Scarsbrook & Townsend, 1993). The four streams in this study all differed in their disturbance regimes; Waingongoro River experienced the most disturbance and Inaha Stream the least disturbed, with Kaupokonui Stream ranking second and Taungatara Stream third. More frequently disturbed streams tended to have a higher species diversity in this study. Waingongoro River had 37 species, and the Kaupokonui Stream had 41, whereas Inaha Stream and Taungatara Stream both had 34 species. As in RG Death (1995), the two more frequently disturbed sites (Waingongoro and Kaupokonui) had similar invertebrate community composition, sharing a number of mayfly, Diptera and Chironomid taxa. These taxa presumably are better adapted to disturbance and can recolonise more rapidly following flooding.

The upstream sites at all rivers were most similar in invertebrate community composition, indicating that these communities were not dealing with the effects of nutrient enrichment, whereas the downstream communities showed marked change in their structure, shifting towards a community more dominated by pollution tolerant species. For the least disturbed sites, Inaha Stream and Taungatara Stream, nutrients were the most important determinant of macroinvertebrate community composition. However, the Waingongoro River and Kaupokonui Stream had similar communities at both upstream and downstream sites, indicating that these invertebrate communities were less affected by nutrient enrichment. This can be seen in Figure 18, where the Waingongoro and Kaupokonui's graph points are closely grouped, indicating a similarity in communities. The Kaupokonui Stream shows a particularly close grouping. This suggests that more frequent disturbance in these streams overrides the effects of nutrient enrichment on the invertebrate community composition. This phenomenon is likely due to the more frequent floods flushing out periphyton and nutrients more regularly and creating a more stable nutrient gradient along the whole catchment.

Future management implications

It is very rare for one variable to act alone in influencing the ecological health of a stream. For stream health to be managed effectively, all the stressors on a stream must be considered together as C. R. Townsend, Uhlmann, and Matthaei (2008) posits. The results of this study illustrate that different stressors may have different formative effects dependant on what other stressors are acting and to what degree. This suggests that a 'one size fits all' approach with regards to nutrient limits may not be the most effective strategy for ecological health. An individualised approach that creates ecological health guidelines for specific rivers of size or importance or for a local area with a number of smaller streams may result in more positive outcomes. The development of indices that disentangle the effects of different stressors in streams would allow prioritisation of management of the most seriously affected catchments. Strategies for future management for the streams included in this study could include higher nutrient allowances for the more flood prone streams to protect ecological health. Taxa richness and abundance in most sites in the tested streams and rivers were generally higher at the upstream sites as compared to the lower reaches, indicating a change in nutrient gradient. However, as discussed above, the two most frequently disturbed streams had a more similar community structure at upstream and downstream sites. This suggests that a variable nutrient criteria could be utilised in these streams with reference to local flood frequency and magnitude to prevent over-proliferation of periphyton as in B. J. Biggs (2000).

Conclusions

There was an overall change in macroinvertebrate community composition downstream as compared to upstream sites. There was a change in the nutrient gradient between upstream and downstream sites at each stream due to landuse changes along the catchment. Flood frequency was different for each of the streams in this study and it appears that more frequent disturbance has the ability to override the negative effects of nutrient enrichment in this study.

Across three of the four waterways sampled, upstream sites with little agriculture in their catchment typically had higher levels of waterway health including biotic and physical indices indicative of good water quality. Invertebrate community composition changed at the downstream sites, downstream sites tended to have invertebrate communities dominated by more pollution tolerant taxa. Periphyton biomass (as measured by Cholorophyll α) increased slightly at downstream sites but did not exceed the Ministry for the Environments recommended guidelines for “clean water” benthic fauna. There was a relationship between phosphorus levels and MCI scores for sites and to a lesser extent with nitrogen also. There was a weak relationship between the Waterway Self-Assessment Score (a measure of physical habitat) and MCI scores but this was not statistically significant.

References

Abell, R. (2002). Conservation biology for the biodiversity crisis: a freshwater follow-up. *Conservation biology, 16*(5), 1435-1437.

Anderson, D. M., Glibert, P. M., & Burkholder, J. M. (2002). Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. *Estuaries, 25*(4), 704-726.

Ballantine, D. J., & Davies-Colley, R. J. (2014). Water quality trends in New Zealand rivers: 1989–2009. *Environmental Monitoring and Assessment, 186*(3), 1939-1950.

Barquín, J., & Death, R. G. (2004). Patterns of invertebrate diversity in streams and freshwater springs in Northern Spain. *Archiv für Hydrobiologie, 161*(3), 329-349.

Barquín, J., & Death, R. G. (2006). Spatial patterns of macroinvertebrate diversity in New Zealand springbrooks and rhithral streams. *Journal of the North American Benthological Society, 25*(4), 768-786.

Bedford, B. L., Walbridge, M. R., & Aldous, A. (1999). Patterns in nutrient availability and plant diversity of temperate North American wetlands. *Ecology, 80*(7), 2151-2169. doi:10.1890/0012-9658(1999)080[2151:pinaap]2.0.co;2

Bertrand, K. N., Whiles, M. R., Gido, K. B., & Murdock, J. N. (2013). Influence of macroconsumers, stream position, and nutrient gradients on invertebrate assemblage development following flooding in intermittent prairie streams. *Hydrobiologia, 714*(1), 169-182. doi:10.1007/s10750-013-1534-5

Biggs, B., Stevenson, R., & Lowe, R. (1998). A habitat matrix conceptual model for stream periphyton. *Archiv für Hydrobiologie, 143*(1), 21-56.

Biggs, B. J. (2000). Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. *Journal of the North American Benthological Society, 19*(1), 17-31.

Boulton, A. J., Peterson, C. G., Grimm, N. B., & Fisher, S. G. (1992). Stability of an aquatic macroinvertebrate community in a multiyear hydrologic disturbance regime. *Ecology, 73*(6), 2192-2207.

Brunsden, D. (2001). A critical assessment of the sensitivity concept in geomorphology. *Catena, 42*(2), 99-123.

Carpenter, S. R., Caraco, N. F., Correll, D. L., Howarth, R. W., Sharpley, A. N., & Smith, V. H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological applications, 8*(3), 559-568.

Chetelat, J., Pick, F., Morin, A., & Hamilton, P. (1999). Periphyton biomass and community composition in rivers of different nutrient status. *Canadian Journal of Fisheries and Aquatic Sciences, 56*(4), 560-569.

Chiew, F. H. S. (2006). Estimation of rainfall elasticity of streamflow in Australia. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques, 51*(4), 613-625. doi:10.1623/hysj.51.4.613

Clapcott, J., Goodwin, E., & Snelder, T. (2013). *Predictive Models of Benthic Macroinvertebrate Metrics. Prepared for Ministry for the Environment*. Retrieved from

Clarke, K. R., & Warwick, R. (2001). Change in marine communities. *An approach to statistical analysis and interpretation*.

Clausen, B., & BIGGS, B. (1997). Relationships between benthic biota and hydrological indices in New Zealand streams. *Freshwater Biology, 38*(2), 327-342.

Connell, J. H. (1978). Diversity in tropical rain forests and coral reefs. *Science, 199*(4335), 1302-1310.

Covich, A. P., Palmer, M. A., & Crowl, T. A. (1999). The Role of Benthic Invertebrate Species in Freshwater Ecosystems: Zoobenthic species influence energy flows and nutrient cycling. *BioScience, 49*(2), 119-127. doi:10.2307/1313537

Cross, W., Wallace, J., Rosemond, A., & Eggert, S. (2006). WHOLE-SYSTEM NUTRIENT ENRICHMENT INCREASES SECONDARY PRODUCTION IN A DETRITUS-BASED ECOSYSTEM. *Ecology, 87*(6), 1556-1565.

Daughney, C., & Wall, M. (2007). Groundwater quality in New Zealand: State and trends 1995-2006. *GNS Science Consultancy Report, 23*.

Davies-Colley, R. J., Nagels, J. W., Smith, R. A., Young, R. G., & Phillips, C. J. (2004). Water quality impact of a dairy cow herd crossing a stream. *New Zealand Journal of Marine and Freshwater Research, 38*(4), 569-576.

Death, R. (1995). Spatial patterns in benthic invertebrate community structure: products of habitat stability or are they habitat specific? *Freshwater Biology, 33*(3), 455-467.

Death, R. (1996). The effect of patch disturbance on stream invertebrate community structure: the influence of disturbance history. *Oecologia, 108*(3), 567-576.

Death, R. (1996). Predicting the impacts of biological and physical disturbances: Does theoretical ecology hold any answers? *New Zealand Journal of Ecology*, 17-26.

Death, R. (2008). The effect of floods on aquatic invertebrate communities. *Aquatic insects: challenges to populations*, 103-121.

Death, R. G., Fuller, I. C., & Macklin, M. G. (2015). Resetting the river template: the potential for climate-related extreme floods to transform river geomorphology and ecology. *Freshwater Biology, 60*(12), 2477-2496. doi:10.1111/fwb.12639

Delong, M. D., Thorp, J. H., Greenwood, K. S., & Miller, M. C. (2001). Responses of consumers and food resources to a high magnitude, unpredicted flood in the upper Mississippi River basin. *River Research and Applications, 17*(3), 217-234.

Demchick, E. (2014). Nutrient enrichment delays flood recovery of stream macroinvertebrate communities.

Dole-Olivier, M. J. (2011). The hyporheic refuge hypothesis reconsidered: a review of hydrological aspects. *Marine and Freshwater Research, 62*(11), 1281-1302. doi:10.1071/mf11084

Dole-Olivier, M. J., Marmonier, P., & Beffy, J. L. (1997). Response of invertebrates to lotic disturbance: is the hyporheic zone a patchy refugium? *Freshwater Biology, 37*(2), 257-276. doi:10.1046/j.1365-2427.1997.00140.x

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z.-I., Knowler, D. J., Lévêque, C., . . . Stiassny, M. L. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews, 81*(2), 163-182.

Dudley Williams, D., & Hynes, H. (1974). The occurrence of benthos deep in the substratum of a stream. *Freshwater Biology, 4*(3), 233-256.

Environment, P. C. f. t. (2013). Water quality in New Zealand: land use and nutrient pollution: Parliamentary Commissioner for the Environment Wellington.

Flecker, A. S., & Feifarek, B. (1994). Disturbance and the temporal variability of invertebrate assemblages in two Andean streams. *Freshwater Biology, 31*(2), 131-142.

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., & Holling, C. S. (2004). Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution, and Systematics*, 557-581.

Foote, K. J., Joy, M. K., & Death, R. G. (2015). New Zealand dairy farming: milking our environment for all its worth. *Environmental Management*, *56*(3), 709-720.

Gabellone, N. A., Claps, M. C., Solari, L. C., & Neschuk, N. C. (2005). Nutrients, Conductivity and Plankton in a Landscape Approach to a Pampean Saline Lowland River (Salado River, Argentina). *Biogeochemistry, 75*(3), 455-477. doi:10.1007/s10533-005-3273-9

Gafner, K., & Robinson, C. T. (2007). Nutrient enrichment influences the responses of stream macroinvertebrates to disturbance. *Journal of the North American Benthological Society, 26*(1), 92-102. doi:10.1899/0887-3593(2007)26[92:neitro]2.0.co;2

Griffiths, G. A., & McKerchar, A. I. (2012). Estimation of mean annual flood in New Zealand. *Journal of Hydrology (New Zealand)*, 111-120.

Harding, J., Young, R. G., Hayes, J., Shearer, K. A., & Stark, J. (1999). Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology, 42*(2), 345-357.

Hauer, F. R., & Stanford, J. A. (1982). ECOLOGICAL RESPONSES OF HYDROPSYCHID CADDISFLIES TO STREAM REGULATION. *Canadian Journal of Fisheries and Aquatic Sciences, 39*(9), 1235-1242.

Holmes Jr, R. R., & Dinicola, K. (2010). *100-Year Flood-It's All About Chance* (2332-354X). Retrieved from

Jacobson, C., Hughey, K. F., Lynch, A., Nursey-Bray, M., O'connell, M., Munro, P., . . . Carter, R. (2014). Twenty years of pacifying responses to environmental management. *Australasian Journal of Environmental Management, 21*(2), 143-174.

Kundzewicz, Z. W., Mata, L. J., Arnell, N. W., Doll, P., Jimenez, B., Miller, K., . . . Shiklomanov, I. (2008). The implications of projected climate change for freshwater resources and their management. *Hydrological Sciences Journal-Journal Des Sciences Hydrologiques, 53*(1), 3-10.

Lake, P. (2000). Disturbance, patchiness, and diversity in streams. *Journal of the North American Benthological Society, 19*(4), 573-592.

Lancaster, J., & Belyea, L. R. (1997). Nested hierarchies and scale-dependence of mechanisms of flow refugium use. *Journal of the North American Benthological Society, 16*(1), 221-238. doi:10.2307/1468253

Larned, S. T., Scarsbrook, M. R., Snelder, T. H., Norton, N. J., & Biggs, B. J. (2004). Water quality in low-elevation streams and rivers of New Zealand: Recent state and trends in contrasting land-cover classes. *New Zealand Journal of Marine and Freshwater Research, 38*(2), 347-366.

Lemly, A. D. (1982). Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. *Hydrobiologia, 87*(3), 229-245. doi:10.1007/bf00007232

Liess, A., Lange, K., Schulz, F., Piggott, J. J., Matthaei, C. D., & Townsend, C. R. (2009). Light, nutrients and grazing interact to determine diatom species richness via changes to productivity, nutrient state and grazer activity. *Journal of Ecology, 97*(2), 326-336.

Matthaei, C. D., Arbuckle, C. J., & Townsend, C. R. (2000). Stable surface stones as refugia for invertebrates during disturbance in a New Zealand stream. *Journal of the North American Benthological Society, 19*(1), 82-93.

Matthaei, C. D., Piggott, J. J., & Townsend, C. R. (2010). Multiple stressors in agricultural streams: interactions among sediment addition, nutrient enrichment and water abstraction. *Journal of Applied Ecology, 47*(3), 639-649.

Matthaei, C. D., Uehlinger, U., Meyer, E. I., & Frutiger, A. (1996). Recolonization by benthic invertebrates after experimental disturbance in a Swiss prealpine river. *Freshwater Biology, 35*(2), 233-248. doi:10.1046/j.1365-2427.1996.00496.x

McDowell, R., Larned, S., & Houlbrooke, D. (2009). Nitrogen and phosphorus in New Zealand streams and rivers: control and impact of eutrophication and the influence of land management. *New Zealand Journal of Marine and Freshwater Research, 43*(4), 985-995.
Milner, A. M., Robertson, A. L., McDermott, M. J., Klaar, M. J., & Brown, L. E. (2013). Major flood disturbance alters river ecosystem evolution. *Nature Climate Change, 3*(2), 137.
Nagels, J., Davies-Colley, R., Donnison, A., & Muirhead, R. (2002). Faecal contamination over flood events in a pastoral agricultural stream in New Zealand. *Water Science and Technology, 45*(12), 45-52.
Nottage, R., Wilcocks, L., Lee, J., Burke, C., Kalaugher, E., Roche, J., . . . Li, F. (2012). Impacts of climate change on land-based sectors and adaptation options: Clark, AJ.
Novotny, V., & Olem, H. (1994). Water quality: prevention, identification, and management of diffuse pollution.
Olsen, D. A., & Townsend, C. R. (2005). Flood effects on invertebrates, sediments and particulate organic matter in the hyporheic zone of a gravel-bed stream. *Freshwater Biology, 50*(5), 839-853.
Pearson, R., & Connolly, N. (2000). Nutrient enhancement, food quality and community dynamics in a tropical rainforest stream. *Freshwater Biology, 43*(1), 31-42.
Peterson, G., Allen, C. R., & Holling, C. S. (1998). Ecological resilience, biodiversity, and scale. *Ecosystems, 1*(1), 6-18.
Pfankuch, D. J. (1975). Stream reach inventory and channel stability evaluation. *US Department of Agriculture Forest Service, Region, 1*.
Polglase, M. A. (2000). Development of the 'Waterway Self-Assessment Form' a stream management tool for landowners *MSc, Massey University*
Poole, W. C., & Stewart, K. W. (1976). The vertical distribution of macrobenthos within the substratum of the Brazos River, Texas. *Hydrobiologia, 50*(2), 151-160.
Quinn, J. M., & Gilliland, B. W. (1989). The Manawatu River Cleanup-Has It Worked? *Transactions of the Institution of Professional Engineers New Zealand: Civil Engineering Section, 16*(1), 22.
Quinn, J. M., & Hickey, C. W. (1990). Magnitude of effects of substrate particle size, recent flooding, and catchment development on benthic invertebrates in 88 New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research, 24*(3), 411-427.
Reice, S. R., Wissmar, R. C., & Naiman, R. J. (1990). Disturbance regimes, resilience, and recovery of animal communities and habitats in lotic ecosystems. *Environmental Management, 14*(5), 647-659.
Resh, V. H., Brown, A. V., Covich, A. P., Gurtz, M. E., Li, H. W., Minshall, G. W., . . . Wissmar, R. C. (1988). The role of disturbance in stream ecology. *Journal of the North American Benthological Society, 7*(4), 433-455.
Riseng, C., Wiley, M., & Stevenson, R. (2004). Hydrologic disturbance and nutrient effects on benthic community structure in midwestern US streams: a covariance structure analysis. *Journal of the North American Benthological Society, 23*(2), 309-326.
Robinson, C. T., Uehlinger, U., & Monaghan, M. T. (2003). Effects of a multi-year experimental flood regime on macroinvertebrates downstream of a reservoir. *Aquatic Sciences, 65*(3), 210-222. doi:10.1007/s00027-003-0663-8
Rosenzweig, M. L. (1995). *Species diversity in space and time*: Cambridge University Press.
Sansom, J., & Renwick, J. A. (2007). Climate change scenarios for New Zealand rainfall. *Journal of Applied Meteorology and Climatology, 46*(5), 573-590.
Scarsbrook, M. R., & Townsend, C. R. (1993). Stream community structure in relation to spatial and temporal variation: a habitat templet study of two contrasting New Zealand streams. *Freshwater Biology, 29*(3), 395-410.
Sims, J., Simard, R., & Joern, B. (1998). Phosphorus loss in agricultural drainage: Historical perspective and current research. *Journal of Environmental Quality, 27*(2), 277-293.
Sponseller, R., Benfield, E., & Valett, H. (2001). Relationships between land use, spatial scale and stream macroinvertebrate communities. *Freshwater Biology, 46*(10), 1409-1424.
Stark, J., Boothroyd, I., Harding, J., Maxted, J., & Scarsbrook, M. (2001). *Protocols for sampling macroinvertebrates in wadeable streams*: Cawthron Institute.

Stark, J. D. (1985). Macroinvertebrate community index of water quality for stony streams.

Stark, J. D. (1993). Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research, 27*(4), 463-478.

Steinman, A. D., Lamberti, G. A., Leavitt, P. R., & Uzarski, D. G. (2017). Biomass and pigments of benthic algae *Methods in Stream Ecology, Volume 1 (Third Edition)* (pp. 223-241): Elsevier.

Stocker, T. (2014). *Climate change 2013: the physical science basis: Working Group I contribution to the Fifth assessment report of the Intergovernmental Panel on Climate Change*: Cambridge University Press.

Suren, A. M., & Jowett, I. G. (2006). Effects of floods versus low flows on invertebrates in a New Zealand gravel-bed river. *Freshwater Biology, 51*(12), 2207-2227.

Townsend, C., Doledec, S., & Scarsbrook, M. (1997). Species traits in relation to temporal and spatial heterogeneity in streams: a test of habitat templet theory. *Freshwater Biology, 37*(2), 367-387.

Townsend, C. R., Uhlmann, S. S., & Matthaei, C. D. (2008). Individual and combined responses of stream ecosystems to multiple stressors. *Journal of Applied Ecology, 45*(6), 1810-1819.

Verburg, P., Hamill, K., Unwin, M., & Abell, J. (2010). Lake water quality in New Zealand 2010: Status and trends: Prepared for the Ministry for the Environment. *Hamilton, New Zealand: National Institute of Water & Atmospheric Research*.

Wallace, J. B., & Webster, J. R. (1996). The role of macroinvertebrates in stream ecosystem function. *Annual review of entomology, 41*(1), 115-139.

Weng, Z., Mookerji, N., & Mazumder, A. (2001). Nutrient-dependent recovery of Atlantic salmon streams from a catastrophic flood. *Canadian Journal of Fisheries and Aquatic Sciences, 58*(8), 1672-1682.

Wharfe, J., Taylor, K., & Montgomery, H. (1984). The growth of Cladophora glomerata in a river receiving sewage effluent. *Water research, 18*(8), 971-979.

Wright-Stow, A., & Winterbourn, M. (2003). How well do New Zealand's stream-monitoring indicators, the macroinvertebrate community index and its quantitative variant, correspond?

Yount, J. D., & Niemi, G. J. (1990). Recovery of lotic communities and ecosystems from disturbance—a narrative review of case studies. *Environmental Management, 14*(5), 547-569.

Zwick, P. (1996). Variable egg development of Dinocras spp (Plecoptera, Perlidae) and the stonefly seed bank theory. *Freshwater Biology, 35*(1), 81-99. doi:10.1046/j.1365-2427.1996.00482.x