Comparison of kōura (*Paranephrops planifrons*) populations in Waikato hill-country streams with varying type and age of riparian planting and fencing

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Abstract

Riparian fencing and planting are commonly used in New Zealand as a tool to mitigate pastoral land-use impacts on streams. *Paranephrops planifrons* (kōura) are an integral and important part of New Zealand stream ecosystems and have been referred to as a keystone species exerting strong influences on stream processes. The overall aim of this research was to establish the influence of riparian planting age on kōura habitat and density, in the context of other environmental (e.g., upstream catchment) and biotic (e.g., eel predators) factors, in 26 small Waikato hill-country stream sites that had catchments in native forest, pasture or established riparian plantings spanning one to 23 years old. Data were collected on site riparian, instream habitat and physicochemical parameters, and kōura and fish were sampled using adapted electric-fishing techniques designed to optimise kōura capture. Geographic Information Systems and available databases were used to characterise upstream catchment characteristics.

Upstream catchment variables were found to exert greater influence on reach-scale (50 m) instream habitat features rather than local riparian conditions or riparian planting age. Important factors explaining variation between sites included upstream catchment size, geology and land-cover. Instream shade followed the expected trajectory of increased canopy cover with riparian age, however, differences between other measures of potential kōura habitat were limited based on riparian planting age classes.

Instream habitat conditions exerted more influence on kōura densities than local riparian conditions, although upstream catchment variables such as catchment size, geology and vegetation featured as predictors for kōura density. Both total and YOY kōura were captured in a range of habitats and were strongly associated with bank features such as root complexes. YOY kōura were influenced by substrate size, hydraulic habitat, and other instream structural features. Riparian planting age did not appear to influence total kōura or YOY density, however, older riparian plantings (>11 years) appeared to function similar to native forest sites, with similar kōura densities.

Fish density appeared to exert some influence on both total and YOY kōura densities, particularly eels ≤300 mm, however, diadromous fish communities in this study were structured primarily by distance downstream to the sea. Eel density was minimally affected by riparian age class and densities were similar in native forest and older riparian sites

This study provided important insights into factors associated with kōura habitat and densities during summer in small, Waikato hill-country streams with different-aged riparian plantings. The overall findings indicate that a wide range of factors at both catchment and reach-scales are influencing kōura densities and persistence in pastoral streams. The lack of an effect of riparian planting age on kōura density partly reflects the wide variability in riparian management between sites (e.g., riparian width, planting density, length). Integration of key catchment and site influences on kōura density established in this study, will help guide planning for future riparian management activities targeting kōura (and eel) population enhancement in pastoral hill-country streams.

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Chapter 1

General Introduction

1.1 Pastoral land-use effects on freshwater ecosystems

Freshwater ecosystems provide a wide range of ecosystem services for humans, including: i) provisioning services such as water for consumption, industry, transport and power generation; ii) regulatory services such as maintenance of water quality, flood buffering and erosion control; iii) cultural services such as recreation and tourism; and iv) a range of supporting services such as nutrient cycling, primary production and ecosystem resilience (Sala *et al.*, 2000; Chopra *et al.*, 2005). In a New Zealand context, freshwater ecosystem services support agriculture, industry, tourism and the health and well-being of people and communities (Ministry for the Environment & Stats NZ, 2017). Additionally, freshwater is highly valued by and integral to the well-being of Māori who view the provision of freshwater ecosystem services in both material (provisioning, regulating and supporting) and non-material terms fulfilling sacred, spiritual and customary-cultural values (Harmsworth & Awatere, 2013).

Increased global human population density, coupled with high levels of economic prosperity, have driven greater development of water resources for agricultural, urban and industrial purposes. As a consequence, water quality has declined globally, and many inland waterways have undergone eutrophication (Chopra *et al.*, 2005).

New Zealand conforms to global trends of freshwater degradation. Recent findings (Ministry for the Environment & Stats NZ, 2019) indicate 71% of river length in pastoral farming areas currently sustain nitrogen levels capable of affecting the growth of sensitive aquatic species and 82% of river length in pastoral farming areas is no longer suitable for swimming based on pathogen levels. Consequently, many New Zealanders now consider water pollution, primarily from pastoral agriculture, as the most important, wide-scale environmental issue affecting the country (Howard-Williams *et al.*, 2010).

Pastoral agriculture is the dominant land-use in New Zealand where it accounts for 40% of land area making production by the primary sector, one of the country's top export earners (Ministry for the Environment, 2007).

Declining water quality in pastoral catchments in New Zealand is driven by nutrients, fine sediments and pathogens (Howard-Williams *et al.*, 2010; Davies-Colley, 2013). Contributions of nutrients such as nitrogen and phosphorus to streams can be a function of: i) access of stock and direct deposition of dung and urine to waterways; ii) nutrients binding to soil particles which are mobilised by soil erosion or stock trampling; and iii) poor fertiliser application and management (Davies-Colley, 2013). Increased sediment can be a function of hillslope erosion; runoff and bank disturbance by livestock (Parkyn & Wilcock, 2004), all of which reduce water clarity (Quinn & Cooper, 1997) and influence nutrient loading. Pathogens are characterised predominantly by faecal microbes such as *Escherichia coli*, and are attributed to runoff of livestock wastes from surrounding land and through direct defecation into streams (Davies-Colley, 2013).

Additionally, pastoral management often involves clearance or reduction in streamside (or riparian) vegetation, thereby influencing the biological and physicochemical nature of streams (Townsend *et al.*, 1997). Implications of riparian vegetation clearance include: i) reduced shade leading to raised stream water temperatures and increased algae and macrophyte growth from higher light penetration (Arnaiz *et al.*, 2011); ii) reduced stream bank stability by removal of root complexes (Belsky *et al.*, 1999); iii) altered energy pathways due to reduced leaf litter and wood inputs with associated changes in trophic structure and species composition (Karr & Schlosser, 1978; Belsky *et al.*, 1999; Giling *et al.*, 2014); iv) reduced sediment retention and increased bank erosion leading to higher sediment loads within streams (Parkyn *et al.*, 2000); v) reduced habitat for aquatic organisms through loss of undercut banks, changes in channel morphology, loss of habitat diversity and reduced habitat stability (Belsky *et al.*, 1999); vi) loss of terrestrial biodiversity associated with streamside vegetation (Naiman & Decamps, 1997); and vii) loss of connectivity and corridors across aquatic and terrestrial landscapes for dispersal and re-colonisation (Ministry for the Environment, 2001).

Stream channels are often modified to increase drainage and assist livestock management (Quinn *et al.*, 1992). Channelisation and straightening are undertaken to improve water conveyance but result in reduced stream habitat diversity and increased bank erosion (Karr & Schlosser, 1978). Removal of wood or other impediments to facilitate flow also simplifies channel morphology and alters instream habitat (Baillie, 2011). Infilling or tiling of small order headwater stream channels, wetlands and seeps results in reduced aquatic habitat area and loss of ecological function which often has a strong influence on the wider health of the catchment (Storey *et al.*, 2009).

Globally, continued demand to increase productivity in agricultural systems will drive ongoing intensification leading to increased impacts on stream ecosystems and further degradation (Yates & Bailey, 2006). Both New Zealand's population and agriculture-based economy are continuing to grow and the expectation is that intensive agriculture will continue to expand into new areas with the potential for greater impacts on water quality in more waterbodies (Ministry for the Environment & Stats NZ, 2017).

In recent decades, rates of land intensification in New Zealand have been among the highest in the world, with increased fertiliser and pesticide use, higher stocking rates and yields, as well as increased stock food inputs and conversion of existing low-intensity pastoral land to more intensive types of agriculture such as dairying (Ministry for the Environment & Stats NZ, 2017). As a result of intensification, water quality and biodiversity have continued to decline in many pastoral catchments (Parliamentary Commissioner for the Environment, 2004).

The continuing decline in water quality and stream biodiversity throughout New Zealand as a result of land-use intensification and the subsequent reduction in its mauri (energy, life force), is a significant issue to Māori (Harmsworth *et al.*, 2016). Manifestation of this decline is usually seen by degradation of customary resources with associated reduction in species abundance and habitat, as well as poor water quality, reduced flows and overall ecosystem decline (Harmsworth *et al.*, 2016).

1.2 Role of riparian zones in mitigating land-use effects

Riparian zones are defined as the area of land immediately adjacent to a stream at the interface of aquatic and terrestrial ecosystems (Gregory *et al.*, 1991). They consist of both abiotic and biotic features, provide both aquatic and terrestrial biological connections across landscapes (Naiman & Decamps, 1997) and perform a range of ecosystem functions (Parkyn *et al.*, 2000). The influence a riparian zone exerts on stream attributes is a function of vegetation type, its location within the catchment and stream width (Naiman & Decamps, 1997; Jowett *et al.*, 2009). Consequently, influences are most pronounced in smaller streams with continuous riparian integrity. There is a vast body of scientific literature on the interactions of riparian zones with stream ecosystems, which is too large to review here. Below I focus on New Zealand examples supported by international publications where appropriate.

In unmodified areas, the riparian zone is usually delineated by the presence of riparian vegetation which comprises an assemblage of species suited to the conditions of the aquatic and terrestrial interface (Ministry for the Environment, 2001). Generally, these zones encompass the area of land from the stream edge to the base of the hills at the extent of the floodplain, and in some steeper catchments part-way up the hillslope (Ministry for the Environment, 2001). The extent and composition of riparian vegetation influences the functioning efficacy and buffering capacity of the riparian zone, while the zone's condition influences the quality of the waterway (Ministry for the Environment, 2001). Additionally, at a catchment-scale, the type and extent of riparian vegetation drives energy and material flows that provide resources for biota within a forest stream ecosystem (Hynes, 1975).

Management of the riparian zone is a tool universally promoted to reduce and mitigate the negative impacts of pastoral land-use activities on stream systems (Parkyn *et al.*, 2000; Greenwood *et al.*, 2012). It is generally applied to the area of land immediately adjacent to a stream and an area where land-use is modified or restricted to reduce adverse effects on stream habitat, water quality and biota (Naiman & Decamps, 1997; Parkyn *et al.*, 2000). Management often consists of a fenced strip alongside a stream which excludes stock and is either left un-grazed or planted with woody vegetation (Parkyn *et al.*, 2000). The width of the managed area can vary depending on the desired management outcomes (Parkyn *et al.*, 2000), however, recommendations can range between 10 and 100 m wide based on stream size, stream order, ecosystem type and adjacent land-use (Allan *et al.*, 1997; Ministry for the Environment, 2001).

Fencing of riparian margins to exclude stock directly mitigates sediment, nutrient and microbial inputs (Wilcock *et al.*, 2013), reduces soil compaction and destabilisation of banks (Belsky *et al.*, 1999), and eliminates direct stock defecation to streams and on stream margins resulting in improvements in microbial water quality (Collins *et al.*, 2007). These outcomes can be particularly pronounced on small streams (Wilcock *et al.*, 2013). Riparian zone planting coupled with fencing can assist in stabilising stream banks and reducing erosion (Naiman & Decamps, 1997), slowing flood flows by increasing the roughness of stream margins (Parkyn *et al.*, 2000), reducing sediment transport from land (Karr & Schlosser, 1978) and mitigating nutrient inflows (Naiman & Decamps, 1997). Fenced riparian zones left in un-grazed pasture can act as filters for particulate contaminants from surface runoff through provision of a dense ground cover which aids in reducing overland flow velocity that facilitates deposition of suspended sediment (Smith, 1989).

Planting fenced riparian zones with shrubs and trees provides shade for streams - a key factor influencing the function of stream ecosystems (Burrell *et al.*, 2014). Shade not only moderates water temperature but also regulates primary productivity by algae (Davies-Colley & Quinn, 1998) and macrophytes (Rutherford *et al.*, 1999; Quinn *et al.*, 2007), with ensuing effects on trophic interactions, community structure and ecosystem metabolism (Quinn *et al.*, 1997). Reduced water temperatures and lower primary production assist in reducing diurnal variation of pH and dissolved oxygen in streams (Wilcock *et al.*, 1999). Provision of shade by riparian trees is most effective for streams less than 6 m wide (Davies-Colley *et al.*, 2009).

Major differences between pasture sites and those with intact riparian vegetation include presence and amount of woody debris, substrate coarseness and water temperature (Hanchet, 1990). Riparian vegetation can influence instream habitat for a range of aquatic species through provision of coarse particulate organic matter (CPOM), root complexes, undercut banks and cover from overhanging stream vegetation (Parkyn *et al.*, 2000; Quinn *et al.*, 2009), which can provide flow refugia and habitat for fish and invertebrates (Johnson *et al.*, 2003). Both small and large wood assist in retention of terrestrial CPOM inputs, a key functional attribute of streams (Quinn *et al.*, 2007). These influences are particularly pronounced for fish and invertebrate communities in small streams (Quinn *et al.*, 1992; Jowett *et al.*, 2009).

Responses of streams to riparian fencing and planting occur over a variety of timeframes. On small, pasture streams at Whatawhata, near Hamilton, pine afforestation created shade close to levels found in native forest streams within 15 years of planting (Quinn *et al.*, 1997). At the same site, reductions in stream water temperature were observed as early as six years after planting in streams 1-2 m wide, however, temperature changes are highly dependent on stream size with smaller streams achieving canopy closure earlier than larger streams (Quinn *et al.*, 1997; Jowett *et al.*, 2009). Collier *et al.* (2001) predicted a reduction of 5-6°C within 15 years in Waikato hill-country streams under a range of riparian management scenarios, although predictions varied depending on channel width and stream size as well as the length of managed areas along with their proximity or connection to other areas of riparian vegetation.

Positive invertebrate community response to riparian planting have been observed over relatively short timeframes in small streams close to headwaters (Quinn *et al.*, 2009). Collier *et al.* (2001) forecast increases in Macroinvertebrate Community Index (MCI) values of 9 to 25% within 15 years of riparian planting of Waikato hill-country streams depending on which areas of the headwater catchment were retired from farming. At the same location, 10 years after stock exclusion and riparian planting, Jowett *et al.* (2009) observed invertebrate metric values in small pastoral streams were only marginally lower than at upstream native sites indicating that upstream connectivity for recolonisation may be important for facilitating recovery. Quinn *et al.* (1997) also observed that the stream invertebrate fauna in the same study area had returned to native forest composition within 15 years of pine afforestation. Although some changes can be observed in relatively short time, full ecological recovery of streams with regard to features such as addition of large wood may take centuries to achieve pre-disturbance conditions (Davies-Colley *et al.*, 2009).

Fish responses to the presence of riparian vegetation have been mixed and difficult to quantify. Rowe *et al.* (2002) found riparian buffer strips generally enhanced native fish communities in steep Coromandel streams draining logged catchments, however, the abundance of eels (*Anguilla* spp.) were not affected. In the Grey River, South Island, Jowett *et al.* (1996) found it difficult to separate the influence of land-use, diadromy and physical habitat on fish distribution and abundance. Holmes *et al.* (2016) assessed the influence of different riparian management regimes on instream habitat condition including fish density and distribution on a Dairy Best Practice monitoring site in South Canterbury. Contrary to expectations, the study found brown trout (*Salmo trutta*) and *Anguilla* spp. evenly distributed throughout the catchment and not strongly correlated with preferred fish habitat such as cover, deep water and low levels of fine sediment, all expected to be enhanced through riparian management.

Responses of fish communities to riparian management can be further complicated by several additional factors. The diadromous nature of most New Zealand fishes causes a natural decline in fish diversity as distance inland and elevation increase (Joy & Death, 2001), so depending on where riparian management occurs fish diversity and abundance may be naturally low. The unknown presence of downstream natural or artificial impediments to fish movement can also influence upstream fish communities, while unknown fishing and harvest impacts can influence abundance and community composition (Holmes *et al.*, 2016). In regards to assessing the influence of riparian planting on fish communities, rehabilitation is often limited to reach scales rather than addressing wider catchment-scale influences and there can be extended lag periods for fish population responses (Holmes *et al.*, 2016).

Davies-Colley (1997) reported that the presence of pasture grasses within the riparian zone of Waikato hill-country streams can directly influence stream morphology through entrainment and stabilisation of sediment on stream banks with effects more pronounced on smaller streams. He found reaches lined by pasture grasses were narrower and more incised than those in native forest with the morphological change becoming evident within a few metres from the pasture-forest boundary. Accordingly, riparian reforestation of pasture streams may cause increased sediment yield as the stream channel changes to a morphology similar to forest following shading out of grasses. Collier *et al.* (2001) forecasted sediment yields from different-sized streams with riparian margins planted in *Pinus radiata* to increase over a period of 25 years until the stream banks reached equilibrium and stabilised.

Further consideration needs to be given to catchment-scale influences on streams when examining the mitigating role of riparian zones. Although the immediate condition of the riparian zone will have a strong influence on some aspects of waterway quality, many influences on streams act at a catchment-scale (Ministry for the Environment, 2001; Parkyn *et al.*, 2003).

Of particular importance is catchment hydrology which is a function of climate, catchment size, underlying geology and topography. Hydrology exerts influence on stream habitat through flow volume, flow variability and water velocity, which interact to affect channel shape, substrate, disturbance regimes, habitat availability, energy transfer and nutrient cycling (Jowett *et al.*, 1996; Allan & Castillo, 2007). In turn, water quality is influenced by catchment geology and hydrology, upstream land-use, land management and vegetation cover which, together, govern nutrient and suspended material loads, and their mobilisation and delivery to receiving waters (Lintern *et al.*, 2018).

The extent that managed riparian areas are connected to each other within a catchment as well as the length and width of managed areas can influence the magnitude of gains in water and habitat quality of downstream reaches (Dodd *et al.*, 2008), and any subsequent response of the macroinvertebrate community (Greenwood *et al.*, 2012). Accordingly, making gains in downstream reaches may be difficult without improving the quality, extent and connectivity of riparian management upstream (Greenwood *et al.*, 2012). In degraded streams, upstream riparian planting may not be enough to improve stream health and other management such as habitat modification may be required, indicating that riparian management should primarily focus on small streams and those with less degraded instream habitat (Greenwood *et al.*, 2012). Indeed, riparian management is often undertaken without an understanding of the scale required to achieve a measurable positive outcome and detect an ecosystem response (Sheldon *et al.*, 2012).

Evaluation of outcomes from riparian management and restoration in New Zealand have, historically, been poor (McKergow *et al.*, 2016). Parkyn *et al.* (2003) found that areas with riparian buffers did show improvement in channel stability and visual water clarity over short time scales, but faecal and nutrient content changes of streams were variable. Changes in macroinvertebrate communities did not appear to be linked to riparian management and the study concluded that their restoration may only be achieved after canopy closure, in areas with long, planted riparian zone lengths and in streams with protected headwaters. The overall success of riparian management is influenced by the spatial arrangement of planted reaches, the size and position of the stream within the catchment, restoration timescales and the proximity of colonists relative to the site undergoing restoration (Parkyn *et al.*, 2003; Wilcock *et al.*, 2009; Greenwood *et al.*, 2012). Upstream influences such as catchment geomorphology and hydrology, along with historic and contemporary land-use, also determine the probability of meeting the desired outcomes of riparian management in a particular stream or catchment (Parkyn *et al.*, 2000; Parkyn *et al.*, 2003; Burrell *et al.*, 2014).

1.3 Riparian management in New Zealand

Large-scale riparian management in New Zealand was first initiated through the Lake Taupō and Upper Kaituna Catchment Control Schemes in the early 1970’s, and was primarily aimed at soil conservation in erosion prone catchments. Studies on the benefits of riparian planting occurred in Taupō (Whangamata Stream), Northland and Nelson during the 1970’s and 1980’s (McKergow *et al.*, 2016). The implementation of the Resource Management Act (RMA) in 1991 marked a turning point in environmental legislation. The Act required sustainable resource use and placed the onus on resource users to avoid, mitigate or remedy any adverse effects on the environment from their activities (Parliamentary Counsel Office, 2018). For rural land owners, the implementation of the RMA was a driving force behind sustainable land and water management. Riparian management was seen as one tool to assist in meeting these obligations and mitigate pastoral land-use effects on streams (Ministry for the Environment, 2001).

The first riparian management guidelines were published in 1995 (Collier *et al.*, 1995), and subsequently various riparian management programmes and research were conducted throughout the 1990's and early 2000's (McKergow *et al.*, 2016). In more recent years, heightened public concern regarding increased degradation of freshwater resources has led to implementation of industry standards (for some sectors) and regulatory tools to mitigate land-use impacts, with riparian management one of the tools promoted to achieve this (Land and Water Forum, 2010). Additionally, current reform of freshwater management in New Zealand (National Policy Statement for Freshwater and National Objectives Framework) may also lead to greater future uptake (McKergow *et al.*, 2016), as well as more stringent and mandatory requirements for sustainable land and water management to meet these new policy requirements.

In New Zealand, riparian management typically consists of fencing the riparian zone to exclude stock, followed by revegetation with native trees and shrubs (Jowett *et al.*, 2009; Howard-Williams & Pickmere, 2010) or exotic grasses, trees and shrubs (Burrell *et al.*, 2014). There are no national standards for a minimum width of managed riparian area with management decisions often based on industry recommendations or accords such as such as the Sustainable Dairying Water Accord (Dairy Environment Leadership Group, 2013).

Measuring the success of riparian management can be problematic due to the inherent variability of streams, particularly due to wider catchment-scale factors and management which can influence results and interpretation of riparian management outcomes (Belsky *et al.*, 1999). Regardless of these limitations, riparian management to mitigate pastoral land-use effects continues to be well utilised in the New Zealand context. For example, under the Waikato Regional Plan 2007, livestock entering or crossing any water body must not reduce the water clarity or increase its suspended solids concentration by more than 10 percent (rule number 4.3.5.4), which generally means that livestock must be prevented from accessing waterways, typically achieved through fencing and bridging (Waikato Regional Council, 2015a). To facilitate uptake of riparian fencing and planting, central government offers income tax deductions for plantings which prevent or mitigate discharges into water courses or waterbodies (Parliamentary Counsel Office, 2007), while financial incentives are offered from the Waikato Regional Council (WRC) for work in priority catchments (Waikato Regional Council, 2019).

Although various studies have assessed the implications of riparian management on macroinvertebrate communities as discussed above, these studies have not specifically focused on responses of freshwater crayfish (*Paranephrops planifrons* or kōura[1]) to implementation of various riparian management techniques, nor assessed the impacts of riparian planting age on kōura response, or the associations of fish on kōura distribution and abundance at riparian sites. These are key information gaps addressed in this thesis.

1.4 Freshwater kōura ecology, values and threats

Two species of endemic freshwater crayfish (kōura) are commonly recognised in New Zealand - *Paranephrops planifrons* and *P. zealandicus*. There may be a third undescribed species occupying areas along the West Coast of the South Island, with indications this group may be more closely related to *P. zealandicus* than *P. planifrons* (Apte *et al.*, 2007). New Zealand freshwater crayfish are commonly known by their Māori name of kōura or kēwai (McDowall, 2011). Both recognised species belong to the Parastacidae family which occurs across a range of geographical areas throughout the southern hemisphere (Chapman & Lewis, 1976). *Paranephrops planifrons* is found throughout the North Island and in the northern and western South Island, while *P. zealandicus* is distributed in the eastern and southern areas of the South Island and on Stewart Island (Hopkins, 1970). Both species occur throughout a wide range of aquatic habitats including lakes, streams and rivers, as well as reservoirs, ponds, swamps and drains (Chapman *et al.*, 2011; Hollows, 2016). This thesis focuses on *P. planifrons* inhabiting streams, although the research outcomes will provide information

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[1] The use of the word kōura throughout this thesis refers to the freshwater crayfish *Paranephrops planifrons* unless otherwise stated.

which may also apply to *P. zealandicus* in relation to riparian management alongside streams.

Estimates vary of kōura longevity, partly because longevity reflects local instream conditions. Hopkins (1967a) found *P. planifrons* did not generally survive beyond three years at study sites in the Wairarapa, while Parkyn *et al.* (2002) found *P. planifrons* in pasture streams in the Waikato reached four years old compared to seven years old in nearby native forest streams.

Breeding in *P. planifrons* occurs between April and December with the gradual onset of breeding and a peak in May and June (Hopkins, 1967b). Female *P. planifrons* reach reproductive maturity between 17 – 20 mm orbit-carapace length (OCL) (Parkyn *et al.*, 2002), and the larger the female, the more eggs are produced (Hopkins, 1967b). Incubation of eggs and young usually takes around 24 weeks and generally lasts until mid-September, with the young passing through two moults after hatching while still attached to the parent. The juveniles leave the mother in January/February. In Waikato hill-country streams, Parkyn (2000) found few *P. planifrons* survived beyond their first year with the highest rates of mortality in winter. Mortality was higher in pasture (39%) than in native forest (18%) streams. Parkyn *et al.* (2002) found females reached the reproductive threshold of 17 - 20 mm OCL within one year in pasture streams in contrast to two years in forest streams where water temperatures were cooler. They observed *P. planifrons* (particularly juveniles) grew faster in pasture streams due to greater moult increments and frequency, as well as faster growth rates.

1.4.1 Trophic and functional roles

Crayfish have the potential to play significant roles in energy transfer in stream and lake food webs through their varied role as detritivores, herbivores and carnivores (Momot, 1995), and through their influence on biomass and species composition of invertebrates and macrophytes (Nystrom *et al.*, 1996). Indeed, *P. planifrons* can act as a keystone species where they occur in high densities through influences on invertebrate community structure directly via predation, or indirectly by influencing organic matter processing rates, and bioturbating benthic sediment (Parkyn *et al.*, 1997; Parkyn *et al.*, 2001). Parkyn *et al.* (2001) found that *P. planifrons* represented a top down predator on invertebrates but were below eels in the trophic hierarchy, while Parkyn *et al.* (1997) suggested that loss of *P. planifrons* from streams could result in an invertebrate community structure alteration, particularly in pool habitats where deposition of leaf matter and fine sediment accumulation are highest.

Diet can show ontogentic shifts between adults and juveniles in some freshwater crayfish species. Thus, juveniles have been found to feed mainly on invertebrates while adults fed mainly on detritus (Guan & Wiles, 1998; Whitmore & Huryn, 1999; Parkyn *et al.*, 2002). *Paranephrops zealandicus* were the dominant shredders contributing to both particulate organic matter production and leaf processing in Otago streams (Usio & Townsend, 2001) where they had indirect effects on stream community structure through bioturbation and excretion (Whitmore & Huryn, 1999). Parkyn *et al.* (2002) found the proportion of each food type consumed by *P. planifrons* differed between land-use which was linked to a variety of factors including habitat choice, available food sources and also likelihood of predator avoidance behaviour.

As previously discussed, improvement of riparian vegetation biodiversity and extent through riparian management provides increased habitat, bank stability, shade and allochthonous inputs to streams. Crayfish play an important role in linking terrestrial and aquatic energy transfer through their ability to process terrestrial leaf litter which is predominantly provided by riparian sources (Jones *et al.*, 2016). The amount of terrestrial input to aquatic systems can therefore have implications for crayfish population success (Jones *et al.*, 2016), and is a function of the type and extent of vegetation within the riparian zone.

If the goal of riparian management is to achieve a functioning stream ecosystem, then a key component will be sustaining kōura populations at sites that have undergone riparian planting.

1.4.2 Vulnerability to predation

Fish community structure has the potential to influence *P. planifrons* distribution, abundance and persistence in stream ecosystems through predation by single or multiple aquatic predators. Vulnerability to predation can be linked to other variables such as gape size of predators (Jones *et al.*, 2016), and habitat complexity influencing the availability of refugia such as coarse substrate and bank cover (Usio & Townsend, 2000). Vulnerability to predation is highest for small crayfish (<15-25 mm OCL) during moult events when their carapace is soft, and when females are gravid (Usio & Townsend, 2000; Jones *et al.*, 2016).

Dominant aquatic predators of *P. planifrons* are brown trout (*Salmo trutta*) (Scott & Duncan, 1967), rainbow trout (*Oncorhynchus mykiss*) (Blair *et al.*, 2012) and eels (*Anguilla australis* and *Anguilla dieffenbachii*) (Hicks, 1997). Olsson et al. (2006) examined the relationship between *P. planifrons* and brown trout in South Island West Coast streams and found the presence of trout influenced crayfish abundance as well as size distribution, with fewer small crayfish being caught in trout streams. Usio and Townsend (2000) and Whitmore and Huryn (1999) also found that adult and young-of-the-year (YOY) *P. zealandicus* were negatively correlated with the presence of brown trout. Blair *et al.* (2012) found rainbow trout (>200 mm) consuming kōura in Lake Rotoiti, central North Island, where they comprised around 6% of their total diet. In Waikato hill-country streams , Parkyn (2000) found eels (*Anguilla* spp.) to be the main predator of *P. planifrons*, particularly on small kōura.

The above studies raise questions concerning the interaction strength of eels (*Anguilla* spp.) with kōura and to what degree this interaction is influenced by land-use which may impose additional stressors such as higher water temperatures, and simplified or unstable refugia that may make kōura more vulnerable to predation. For example, comparison of predator densities between pasture and forested stream sites in the Hakarimata Ranges, Waikato, revealed that between 50-80% of pools in pastoral sites were inhabited by large or medium-sized eels compared to 10-20% of forested study sites (Hanchet, 1990). In that study, biomass of eels averaged 8.6 kg in the two pastoral sites sampled compared to 1.5 kg in adjacent forested sites. The authors concluded that predation by a higher biomass of eels in pasture sites may affect prey community structure. Although *P. planifrons* were not specifically mentioned in that study, these findings are likely relevant to predation of kōura in pasture streams where eels are present.

Another study of native fish and *P. planifrons* in Waikato hill-country streams by Hicks and McCaughan (1997) found a seven-fold increase in eel biomass in pastoral streams compared with forested streams. They concluded that changes in the fish community were likely caused by increased temperature, light, inorganic nitrogen concentrations and in-stream primary production caused by removal of forest cover from pasture streams. Increased eel abundance in response to such factors could have implications for predation rates on *P. planifrons*. Glova *et al.* (1998) found medium to large shortfin eels (≥ 300 mm length) strongly associated with overhead riparian cover, while longfin eels associated with in-stream debris, macrophytes and bank cover. While riparian planting may play a role in providing additional refugia from predation for kōura, it may also influence abiotic and biotic factors affecting abundance of both eel and trout species in pasture streams.

The singular or combined impact of these predatory species on kōura distribution and abundance in streams with different-aged riparian planting has not previously been examined. Such relationships are important to understand in restoration activities if the goal is to achieve sustainable and enduring populations of kōura at managed sites.

1.4.3 Habitat preferences

Physical habitat

Kōura are primarily nocturnal and generally seek cover during the day (Collier *et al.*, 1997). Preferred habitat reported for *P. planifrons* are tree roots, leaf litter, undercut banks, and fallen logs/woody debris, particularly where this material from riparian sources is associated with banks alongside pools or forms debris clusters (Parkyn & Collier, 2004; Jowett *et al.*, 2008). *Paranephrops planifrons* abundance also appears to be influenced by substrate size, with a preference for cobbles rather than sand and boulders (Olsson *et al.*, 2006). Similarly, Jowett *et al.* (2008) found *P. planifrons* abundance reduced as substrates became larger (e.g. dominated by boulders) and as streams became wider than 6 m. In pasture streams, where riparian tree cover is often lacking, kōura commonly use cobbles and instream vegetation such as grass and macrophytes (Parkyn, 2000) although they can also be found in high densities in areas with softer substrates such as clay where tunnelling and burrowing is possible (Riordan, 2000; Hollows, 2016). Size dominance hierarchies are known to occur in freshwater crayfish and relate to available refugia as well as food resources (Bergman & Moore, 2003), with antagonistic behaviour reduced when adequate refugia are provided (Stewart & Tabak, 2011).

Paranephrops planifrons is typically associated with slow-flowing or still-water habitats where velocities are <0.4 m/s (Jowett *et al.*, 2008). They are known to utilise low flow refugia amongst stream substrate and cover at flow velocities of up to 1.6 m/s, with stream edge cover being particularly important as refugia for YOY kōura (Jowett *et al.*, 2008). The ability of *P. planifrons* to persist during high flow disturbance has been strongly linked to the availability of spatial refugia providing habitat stability associated with riparian edges where tree roots, pools and undercut banks occur. A study in hill-country streams at Whatawhata demonstrated that the majority of refugia in pasture streams were cobbles and macrophytes that proved unstable during a large flood event, resulting in significant negative long-term impacts on the kōura population (Parkyn & Collier, 2004). At a larger scale, March and Robson (2006) found patches of forest or vegetated areas within agricultural catchments can provide stable habitat for crayfish from high flow events and provide a source for repopulation to downstream areas.

Riparian planting and management of pasture streams has been noted to increase variables which are key habitats for kōura and which may provide more stable high flow refugia than pasture alone (Parkyn, 2000; Quinn *et al.*, 2009). Additionally, different types of riparian plants, plant age, and position of the planted reach in a catchment may offer different types of refugia at different times during their establishment. Indeed, Burskey and Simon (2010) found important predictors of presence of ten crayfish species of four genera (*Cambarus, Fallicambarus, Oronectes* and *Procambarus*) in west-central Indiana streams were reach-scale variables associated with maintenance of instream cover and intact riparian areas.

Physicochemical conditions

Physicochemical variables can influence the distribution and persistence of crayfish populations within aquatic ecosystems (Jones *et al.*, 2016). Variables such as conductivity, pH, magnesium, calcium, potassium and sodium were all found to influence *Austropotamobius pallipes* presence in British streams (Smith *et al.*, 1996). Calcium is of particular importance during the exoskeleton moulting process associated with crayfish growth and is sourced both from internal stores (gastroliths and consumption of the shed skin) as well as from surrounding water (Taugbol *et al.*, 1996). Hammond *et al.* (2006) found the higher the concentration of calcium in the water, the less time taken for exoskeleton hardening after moulting with an associated reduction in predation vulnerability. The speed at which the new exoskeleton hardens after moulting is therefore key to limiting vulnerability to both cannibalism and predation during moulting (Hollows, 2016). However, Olsson *et al.* (2006) found *P. planifrons* in streams with calcium concentrations as low as 0.9 mg/L with pH ranging between 4.1 and 7.9, indicating high dissolved calcium is not a pre-requisite for crayfish presence. For aquaculture of *Paranephrops* spp. it has been suggested that holding waters contain a minimum concentration of >5 mg/L of calcium with pH in the range of 7-8.5 (Hollows, 2016).

Temperature is the primary determinant of growth in *P. planifrons* (Parkyn *et al.*, 2002) and also influences crayfish movement and dispersal (Jones *et al.*, 2016). This species appeared to be more mobile during summer in studies undertaken in the Waikato, and movement was greater in pasture streams compared with forest streams suggesting activity moderated by water temperature (Parkyn, 2000). Thermal tolerances for survival of *P. planifrons* are relatively high, however, in an experiment to simulate exposure to high temperatures in pasture streams, 100% of crayfish exposed to 26.2°C survived after 24 hours compared to 5-10% at 28.9°C and 0% at 32.4°C. When hypoxia stress was added to five days of exposure to 26.2°C (to simulate deoxygenation in eutrophic lowland streams) survival decreased to 40%, compared to 95% in crayfish held at 19°C (Albert *et al.*, 2015). During studies of acute lethality of low dissolved oxygen, *P. planifrons* held for 48 hours at 15°C showed a LC_{50} of 0.77 ± 0.06 mg/L, suggesting tolerance for low dissolved oxygen conditions (Landman *et al.*, 2005). Animals in the study were denied access to the water surface and were unable to leave the tanks so tolerance may be lower if access to such additional oxygen sources had been available.

The influence of suspended solids on kōura is poorly understood, however, Usio and Townsend (2000) found a negative association between *P. zealandicus* and suspended solids in eastern Otago. Jones *et al.* (2016) make the point that all crayfish bioturbate to some extent so must have some tolerance to short periods of elevated sedimentation.

Physicochemical parameters of streams can be influenced by land-use, catchment geology and the condition of the riparian zone. As previously discussed, riparian management can strongly influence many of these parameters, particularly through the provision of shade and the mitigation of sediment and nutrient inputs to streams. Smith *et al.* (1996) found that, if water chemistry was suitable for crayfish presence, the abundance of *Austropotamobius pallipes* was primarily determined by bank gradient, presence of riparian trees and shrubs, and by the extension of tree roots into water. This finding suggests that, if physicochemical parameters are suitable for kōura, then their abundance may be influenced by other physical habitat parameters such as availability of refugia which can increase as an outcome of riparian management.

1.4.4 Anthropogenic threats

Degradation of freshwater ecosystems both globally and within New Zealand place pressure on freshwater crayfish through habitat loss, effects of land management practices, water abstraction and related flow control, and introduced predators or competitors (Souty-Grosset, 2016; Williams *et al.*, 2017). These threats can occur in isolation, or, more commonly, are cumulative and may act synergistically to exacerbate population decline, eventually becoming critical for population persistence (Richman *et al.*, 2015).

Land-use changes can alter crayfish diet and their trophic interactions with other species (Giling *et al.*, 2009), while the clearance of riparian vegetation often removes key habitat, cover and refugia for invertebrates, increases stream temperature (Jowett *et al.*, 2009) and reduces detritus inputs, bank stability and water quality (Quinn *et al.*, 1992). Additionally, natural or man-made barriers and modified flows can impede crayfish movement within or between catchments (Jones *et al.*, 2016), while introduced fish species or changes in native fish abundance may influence *P. planifrons* populations through predation and habitat modification (Clearwater *et al.*, 2014).

Eutrophication of waterways can modify water chemistry parameters which are increasingly identified as threats to crayfish populations (Richman *et al.*, 2015), including long term exposure to nutrients such as nitrite, nitrate and ammonium which can reduce crayfish immunity and increase vulnerability to disease and parasite infection (Meade & Watts, 1995). Diffuse inputs such as those from agricultural land-use can contain copper, zinc and cadmium as well as various metals, metalloids and chemicals present in feed supplements, fertiliser and pesticides used in pastoral farming, all of which have the ability to be toxic to aquatic life via interactions with gill tissues and dietary uptake with consequences to metabolic processes (Collier *et al.*, 2017).

Although riparian management as a pastoral land-use mitigation tool can reduce the impacts of some of these threats, many operate at catchment-scales and are unlikely to be mitigated by riparian management alone. In a study of reach and catchment-scale factors influencing crayfish in streams in Indiana, USA, Burskey and Simon (2010) reported that instream cover was the most important predictor at a reach-scale, while intact vegetated riparian zones and forested upstream land were most important at a catchment-scale. This contrast indicates the importance of maintaining intact riparian areas to sustain instream cover and crayfish population persistence.

1.4.5 Cultural value

The term cultural keystone species (CKS) is used to describe an animal which can influence the cultural identity of a group through its use for celebrations, ceremonies and as a traditional food (Noble *et al.*, 2016). Kōura is one such species for Māori (Kusabs *et al.*, 2015; Noble *et al.*, 2016) as it was once an important food source. This was particularly so in some inland districts of the central North Island where they were historically found in large numbers (Kusabs

et al., 2015), and were prized by both local tribes and visitors as well as being a tradable commodity around various districts (McDowall, 2011).
Kōura is still valued for customary fishing by Māori in various areas throughout New Zealand (Kusabs *et al.*, 2015; Noble *et al.*, 2016). They are regarded as a taonga (highly prized item), are linked to overall Māori wellbeing, and are part of Māori traditional practice and knowledge (mātauranga) and the enactment of kaitiakitanga (guardianship/stewardship) of natural resources (National Institute of Water and Atmospheric Research, 2016).

When surveyed for the Waikato River Independent Scoping Study, all five Waikato River iwi (tribes) noted a major decline in the abundance and availability of kōura in the Waikato and Waipā catchments to the extent they are no longer common (National Institute of Water and Atmospheric Research, 2010). Nationally, many Māori believe kōura and other CKS species have declined to a point where their current state is no longer sufficient to meet their cultural, economic and social viewpoints (National Institute of Water and Atmospheric Research, 2016). Ongoing management of CKS such as kōura can play an important role in maintaining connections to traditional practices (Noble *et al.*, 2016).

1.5 Objectives and outline of thesis

As outlined above, kōura are an important component of New Zealand stream ecosystems (Parkyn *et al.*, 2001) and riparian fencing and planting are a key management tool used in New Zealand to mitigate the effects of pastoral land-use on streams (McKergow *et al.*, 2016). However, the influence of riparian restoration and management on kōura distribution and abundance in pastoral streams has not been tested. Accordingly, the overall aim of this research is to provide key information for environmental managers and kaitiaki on the response of kōura populations to riparian planting and fencing along pastoral streams. I use a ‘space for time’ approach to evaluate the effects of riparian planting age, and conduct Geographic Information System (GIS) analyses to derive catchment-scale variables that may moderate response of kōura and associated fish populations to riparian planting. Findings from this research will inform riparian management options by providing guidance on important variables associated with *P planifrons* population recovery and enhancement in managed streams.

Specifically, the following objectives developed for this study conducted in 26 Waikato hill-country streams were to:

- Determine the relationship between *P. planifrons* populations (abundance and size structure) and riparian planting of different ages;
- Identify key instream habitat factors affecting kōura populations, and the association of these with riparian planting age;
- Investigate the influence of upstream catchment characteristics on *P. planifrons* populations (distribution and abundance), in particular the degree of connectivity to indigenous forest, forest remnants and/or upstream riparian retirement;
- Determine the association between fish communities and *P. planifrons* populations, with a particular focus on potential predators, in sites with different riparian planting ages and upstream catchment conditions.

This thesis comprises six chapters with three main results chapters. Chapter 1 provides background information on the effects of pastoral land-use on streams, riparian management as a tool to mitigate land-use impacts on streams, and kōura ecology in New Zealand with some relevant global references. Chapter 2 describes the general study area and discusses the site identification and selection process. Chapter 3 examines the linkages between upstream catchment and riparian characteristics to instream habitat variables at the reach-scale. To do this, I used existing spatial databases and conducted GIS analyses to derive relevant catchment-scale variables, and conducted assessment of riparian planting characteristics and instream habitat features. Chapter 4 examines the relative influences of upstream catchment, riparian and instream habitat variables on *P. planifrons* abundance and size structure. Chapter 5 identifies the associations between fish communities and *P. planifrons* populations within sample reaches in relation to upstream catchment, instream habitat and riparian variables, with a particular emphasis on eels as a potential key predator, and explores factors affecting eel density. Chapters 4 and 5 involved a novel adaptation of electric-fishing methods to collect fish and kōura. The final discussion chapter (Chapter 6) reviews and summaries the findings from Chapters 3, 4 and 5, and provides recommendations for key considerations if managing riparian zones for the purposes of kōura enhancement.

Chapter 2

Study area and site selection

2.1 Study area characteristics

The study was carried out at 26 sites in hill-country streams of the western Waikato region, central North Island, where the landscape is dominated by the basaltic volcanos of Karioi and Pirongia and the ranges of the Waipā Fault Scarp which divide the Hamilton basin from the Raglan hills (Selby & Lowe, 1992). These volcanoes and much of the hill-country still have upper slopes in native forest surrounded by farmland on the lower slopes. Streams draining these volcanoes and ranges flow directly to the coast or into the Waipā River catchment – a tributary of the Waikato River. This study focussed on three areas or spatial clusters within the western Waikato region, all within 50 km of Hamilton: i) Pirongia; ii) Whaingaroa; and iii) Whatawhata (Figure 2.1).

2.1.1 Topography and land-use

Mount Pirongia rises to 959 m above sea level (Department of Conservation, 2014) with the upper mountain fully forested and encompassed within Pirongia Forest Park (Figure 2.1). The eastern and southern flanks of the mountain, where sampling sites were located, flow to the Waipā River and are a mix of dairy and dry stock pastoral land use.

Mount Karioi dominates the topography around Whaingaroa and rises to 765 m above sea level (Department of Conservation, 2014) immediately behind Raglan township (Figure 2.1). The majority of the mountain is fully forested and encompassed within a north-western section of Pirongia Forest Park. The remainder of the catchment varies from steep areas to fertile river valleys. Historically, pastoral land-use was dominated by dry stock, however, dairy farming has increased in recent years (Environment Waikato, 2002a). The streams examined in this study enter Whaingaroa harbour apart from one draining directly into the sea from the south-eastern slopes of Karioi.

Whatawhata is characterised by steep and undulating topography (Quinn *et al.*, 2009) with altitude ranging from 50 to 375 m above sea level (Figure 2.1). The area was originally covered in mixed broadleaf-podocarp forest which was cleared in the 1920's and converted to pastoral farming (Dodd *et al.*, 2008). Tracts of native forest still exist in some tributaries with a portion of the catchment within Kakariki Scenic Reserve and a portion in plantation forestry. There are various small native forest remnants throughout the area. Streams in the Whatawhata area drain into the Waipā River.

2.1.2 Climate

The study area is generally characterised by relatively warm temperatures in summer, ranging between 20 and 25°C mean daily maxima, and relatively cold winter temperatures between 0 and 8°C mean daily maxima (Chappell, 2013). The common wind direction throughout the Waikato is northerly or westerly air flow. Average annual wind speed of 11 km/h and average annual rainfall of 1072 mm have been recorded in Hamilton (Chappell, 2013), c.15-45 km to the east of the study sites. The present study was conducted over November to January during which 26% of Hamilton's annual average rainfall generally occurs (Chappell, 2013).

2.1.3 Geology and soils

The New Zealand Land Resource Inventory (NZLRI)[2] was used to establish dominant geology (top rock) for the study sites. Sites located around Whatawhata are dominated by greywacke while sites around Whaingaroa and Pirongia are dominated by rock of volcanic origin, and included volcanic lavas and rock generated from ashes older than the Taupō eruption. Soils through Whaingaroa and Pirongia are primarily volcanic loamy clays which are moderately free draining and suitable for pastoral farming (Molloy, 1998). Whatawhata is dominated by yellow brown soils (Quinn *et al.*, 2009) which are moderately suitable for pastoral use but have weakly developed soil structure and can be prone to erosion (New Zealand Society of Soil Science, 1974).

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[2] Data reproduced with the permission of Landcare New Zealand Limited

Geology is similar within each cluster of sites with the exception of KA1 located near Limeworks Loop Road (see Figure 2.1) where the geology includes greywacke, volcanic rock, sandstone and unconsolidated sands and gravels.

2.1.4 Stream water quality

In general, the water quality of the Waipā River declines from upper to lower reaches, with high sediment inputs from unstable soils and erosion of stream banks. Ongoing change of land-use throughout the catchment from hill country farming to dairy farming is increasing nutrient levels and faecal contamination (*Escherichia coli*) (Waikato Regional Council, 2014). Consistent with national trends, tributary streams in the upper-reaches of catchments with forested headwaters remain of good quality, however, water quality quickly declines as streams flow into lowland areas (Parliamentary Commissioner for the Environment, 2004).

Many of the small streams draining from Karioi flow directly to Whaingaroa harbour. Although water quality in and close to forested headwater catchments is clear and cool, the cumulative effects of extensive and intensive farming and inherently unstable soils result in many of the larger, receiving rivers being silty and nutrient-enriched with relatively poor water quality (Environment Waikato, 2002a).

2.1.5 Fish communities

Fish communities can be influenced by a range of factors. For the study area, the main influences on community structure are likely to be: i) distance to the coast and altitude of sampling sites influencing fish diversity; ii) the presence of barriers which may inhibit migratory processes and regulate fish diversity (David *et al.*, 2017); and iii) water quality (Ministry for the Environment & Stats NZ, 2017).

Fish present in the Karioi area include longfin eel (*Anguilla dieffenbachii*), shortfin eel (*Anguilla australis*), red fin bully (*Gobiomorphus huttoni*) and banded kōkopu (*Galaxias fasciatus*), while streams around Pirongia contain Crans bully (*Gobiomorphus basalis*), short and longfin eels, torrentfish (*Cheimarrichthys fosteri*), shortjaw kōkopu (*Galaxias postvectis*) and brown and rainbow trout

(*Salmo trutta* and *Oncorhynchus mykiss*) (Waikato Regional Council, 2015b). Fish communities in Whatawhata streams consist of long and shortfin eel, Crans bully, lamprey (*Geotria australis*), common smelt (*Retropinna retropinna*), banded kōkopu, torrentfish and giant kokopu (*Galaxias argenteus*) (Hicks, 2003).

2.2 Riparian management

Although riparian management has been extensively applied at pastoral sites in the Waikato, many projects are small in scale, have varying levels of connectivity to other riparian areas, are at different locations within catchments, and have involved a variety of management techniques. These factors posed some challenges in locating suitable sites that are comparable, in terms of different-aged riparian plantings with fencing, and with similar stream size, flow, and other reach and catchment characteristics.

Sites around Pirongia and Whatawhata have been a focus of riparian retirement through the Waikato Regional Council (WRC) Clean Streams programme from 2000-2010 (Campbell, 2002), and more recently through WRC Project Watershed funding (Environment Waikato, 2002b) and through the Waipā Catchment Plan (Waikato Regional Council, 2014). The Whaingaroa catchment has undergone substantial riparian retirement on inflowing streams and harbour margins since 1995 (Environment Waikato, 2002a) and continues to be a focus for catchment works highlighted within the West Coast Zone Plan (Waikato Regional Council, 2016). In the Toreparu catchment, the stream headwaters begin on the eastern flanks of Mount Karioi and drain to the coast between Whaingaroa to the north and Aotea Harbour to the south. The focus of riparian works there has been on fencing and protection of the 223 ha wetland at the base of the catchment (Bourke, 2019).

The former AgResearch Whatawhata Research Centre farm and the surrounding area have been the subject of extensive research on riparian-stream interactions since the 1990's (Quinn & Cooper, 1997). A focus of this research has been investigating the effects of integrated catchment management for a model sheep and beef hill farm on stream water quality and ecology (Quinn, 2009). In more recent times, Whatawhata has been a focus for implementation of riparian management to achieve outcomes within the Waipā Catchment Plan (Waikato Regional Council, 2014), and in a wider context to achieve the Vision and Strategy for the Waikato and Waipā Rivers – Te Ture Whaimana o Te Awa o Waikato (Waikato River Authority, 2010).

2.3 Site screening and selection process

2.3.1 Site screening

Identification of candidate riparian sites

A central aim of the present study was to examine the influence of riparian management on kōura populations. Identification of potential sampling sites focused on catchments which have been both historic and recent focal areas for riparian management as discussed above, are located in similar geographic areas, and have riparian plantings and retirement of different ages. Additionally, study sites were not to be influenced by downstream dams which would prevent recruitment of eels and other migratory fish.

A desk top Geographical Information System (GIS) process was used to identify potential sites in the three focal areas that met the above criteria and could provide a range of different-aged riparian plantings. Information on riparian enhancement work funded by WRC is recorded on the Council's asset management system available in a GIS layer providing spatial extent and age of works. A data agreement was formulated to access this information for site screening purposes using: i) "River and Catchment Services Clean Streams Assets"; and ii) "River and Catchment Services Soil Conservation Assets" GIS layers. The WRC 2012 and 2017 WRC Waikato Region Aerial Photography Survey (WRAPS) layers and NZTM 2000 topographic map layers were accessed through creative commons.

The locations of WRC funded riparian works were plotted and examined in QGIS (v2.18, Open Source software, Switzerland) to screen sites that were: i) a minimum length of 200 m riparian retirement of a particular age; ii) a minimum distance between other candidate sites of 500 m if located on the same tributary; iii) stream order of 1-3; and iv) considered to be wadeable with at least 90% of the sample reach 0.6 m deep or less (Joy *et al.*, 2013) by examination of aerial photography.

These criteria were applied to select smaller streams that were more likely to be influenced by riparian planting and could be sampled effectively. Sample reaches were embedded in a longer riparian strip to enable upstream riparian effects to have some influence, yet far enough apart to provide some level of assumed site independence.

Determining presence of kōura

For candidate sites identified in the initial GIS scoping exercise, the presence of kōura in the catchments of interest was investigated. The New Zealand Freshwater Fish Database was accessed on 2nd August 2018 and records for kōura presence were downloaded for the lower Waipā River and associated tributaries, the Toreparu catchment and the Whaingaroa catchment. To provide relatively recent information all records prior to 2000 were removed from consideration in the dataset.

In addition, WRC data for freshwater fish and invertebrate monitoring was obtained to provide additional kōura location information. The locations of all kōura records were plotted using QGIS and overlaid with known riparian planting sites. Site selection focused on sites in catchments with known presence of kōura, however, data were limited and some sites were chosen based solely on riparian age in the absence of confirmed kōura presence.

2.3.2 Site selection

Using the desktop study results of candidate sites, a short list was compiled and discussed with relevant WRC Catchment Management Officers to gather additional information on the extent and type of riparian planting at each site and general site characteristics. An amended list was used to undertake wider site reconnaissance that involved viewing sites and catchments from public roads to assess site suitability and also looking for sites outside the scope of the WRC work programmes.

Sites identified and confirmed during the site reconnaissance were provided to WRC Catchment Management Officers who made contact with many landowners to seek permission for staff to pass on their contact details. For those who did support the work, contact was made and an information sheet outlining the work provided. For sites identified that were not part of WRC work programmes, landowners were approached in person and project information supplied. Human ethics approval was obtained from The University of Waikato to discuss and gather information on riparian and fencing age from landowners.

Site visits were made to confirm site suitability, and select a suitable 50-m long sample reach that contained a variety of representative habitats and was at least 100 m downstream of the upstream extent of the retired area. The sample reach also needed to be wadeable for electric-fishing with adequate access from the banks. If required, the sample site was moved further downstream to fulfil these criteria.

Additional sites in each of the three spatial clusters were selected to represent pasture reaches with no riparian management and reference reaches with native forest in the entire upstream catchment. Native study sites were located on public conservation land and did not have any fencing as they were surrounded by contiguous native vegetation. Pasture sites also had no fencing and streams were fully accessible to stock.

With assistance of the National Institute of Water and Atmosphere’s (NIWA) Māori Environmental Science Team (Te Kuwaha), information on all potential sites was provided to iwi with an interest in the study area and feedback on specific site locations requested. After iwi feedback was obtained, a Research and Collection permit application for study sites located on land administered by the Department of Conservation was lodged and granted (70894-FAU). No concerns were raised by iwi for any other potential sites.

2.4 Study sites

The locations of study sites selected following the procedures described above are illustrated in Figure 2.1 with selected photos in Figure 2.2, Figure 2.3 and Figure 2.4. Site location and riparian characteristics are shown in Table 2.1 and Table 2.2, respectively. All sample reaches were 50 m in length except for BC1 (43.5 m) which had to be shortened due to some large boulder areas in the sample reach which were unable to be electric-fished effectively. All other data (riparian and instream variables) were collected for the 50 m reach at this site.

The majority of sites were located within pastoral land-use, however, several sites had partially been changed to other land-use post-riparian planting. BC2 land-use on the true left bank (TLB) has recently been changed to an industrial area, while AA1 TLB was bounded by a residential road and TA1 true right bank (TRB) was bounded by a residential property.

Of the 20 riparian sites sampled, eight were recent (≤ 5 years old), three were mid-age (6-10 years old), five were mid-late age (11-15 years old) and four were older (>16 years old) plantings. Three pasture sites with no fencing or planting were sampled to represent zero riparian treatment. Three native forested sites were sampled to represent a reference condition to establish where in the trajectory of restoration the riparian sites were placed based on age.

The Pirongia cluster had five sites ≤ 5 years old, three sites 6-10 years old, one site 11-16 years old and zero sites >16 years old. The Whaingaroa cluster had one site ≤ 5 years old, zero sites 6-10 years old, four sites 11-15 years old and two sites >16 years old, while Whatawhata had two sites ≤ 5 years old, zero sites 6-10 years old, zero sites 11-15 years old and two sites >16 years old (Table 2.2). In addition, all spatial clusters included one pasture site and one native site.

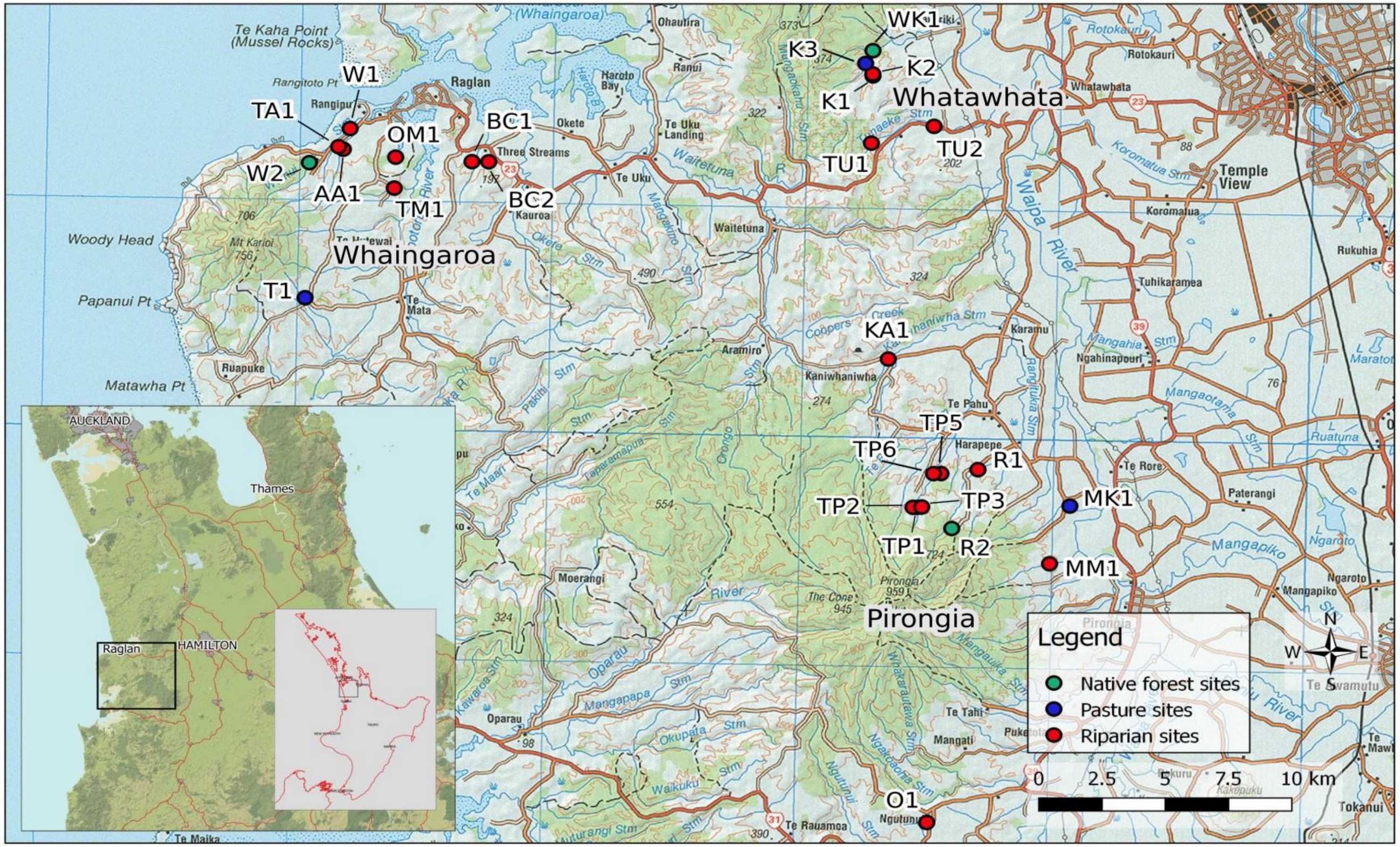

Figure 2.1: Location of study sites around Whaingaroa (n = 9; top left), Pirongia (n = 11; centre and lower), and Whatawhata (n = 6; top centre) in the western Waikato region, North Island, New Zealand (see inserts).

Table 2.1: Location information for all sampling sites (n = 26). Site spatial cluster: PRG = Pirongia; WG = Whaingaroa; WW = Whatawhata.

Site code	**Site spatial cluster**	**Stream name**	**Catchment**	**NZTM Easting upstream end of sample reach**	**NZTM Northing upstream end of sample reach**
KA1	PRG	Kauri	Waipā	1783429	5803408
R1	PRG	Rangitukia	Waipā	1787065	5798650
TP3	PRG	Te Pahu Trib	Waipā	1784866	5796968
TP5	PRG	Te Pahu Trib	Waipā	1785600	5798443
TP6	PRG	Te Pahu Trib	Waipā	1785300	5798429
MM1	PRG	Mangamauku	Waipā	1789960	5794576
O1	PRG	Ongarara	Waipā	1785304	5783184
TP2	PRG	Te Pahu Trib	Waipā	1784523	5796947
TP1	PRG	Te Pahu Trib	Waipā	1784694	5796955
R2	PRG	Rangitukia	Waipā	1786101	5796053
MK1	PRG	Mangakara	Waipā	1790711	5797113
BC2	WG	Bridal Creek	Whaingaroa	1767495	5811751
AA1	WG	Ahi Awa	Whaingaroa	1761662	5812249
BC1	WG	Bridal Creek	Whaingaroa	1766859	5811757
OM1	WG	Omahina	Whaingaroa	1763806	5811898
TA1	WG	Te Aewa	Whaingaroa	1761543	5812304
TM1	WG	Tamoremahoe	Whaingaroa	1763781	5810537
W1	WG	Wainui	Whaingaroa	1762014	5813117
W2	WG	Wainui	Whaingaroa	1760400	5811589
T1	WG	Toreparu	Toreparu	1760344	5805684
TU2	WW	Tunaeke	Waipā	1785032	5813599
TU1	WW	Tunaeke	Waipā	1782592	5812820
K1	WW	Kiripaka Trib	Waipā	1782614	5815767
K2	WW	Kiripaka	Waipā	1782587	5815887
WK1	WW	Whakakai	Waipā	1782589	5816860
K3	WW	Kiripaka	Waipā	1782307	5816301

Figure 2.2: Photos of a selection of sites in the Pirongia spatial cluster.

A = MK1; B = R1; C = TP2; D = TP6

A B

C D

Figure 2.3: Photos of a selection of sites in the Whaingaroa spatial cluster.

A = T1; B = W1; C = TM1, D = BC1.

Figure 2.4: Photos of a selection of sites in the Whatawhata spatial cluster.

A = K2; B = K3; C=TU2; D = K1

Table 2.2: Riparian characteristics and sampling date of 26 study sites. Type of site: Recent = riparian planting 1-5 years; Mid = riparian planting 6-10 years; Mid-late = riparian planting 11-15 years; Old = riparian planting 16+ years, Native = fully native forested catchment; Pasture = no riparian fencing or planting. Fencing type on true-left (TLB) and true-right (TRB) banks: W = wire, PB = post and batten, E = electric, NF = no fence. Average fence distance = average distance (n=6 for each sample reach) taken from centre of stream to fence at edge of riparian buffer. N/A = not applicable.

Site code	**Date sampled**	**Riparian planting age group**	**Age of planting**	**Age of fencing**	**Fencing type (TLB)**	**Fencing type (TRB)**	**Average fence distance (m)**
MK1	29/11/2018	Pasture	0	0	NF	NF	0
KA1	22/01/2019	Recent	3	3	8WPB+1E	8WPB	6.2
R1	13/11/2018	Recent	2	2	5W+2E	5W+2E	6.1
TP3	23/11/2018	Recent	2	2	4W+2E	4W+2E	8.3
TP5	16/11/2018	Recent	4	4	3WE	3WE	10.8
TP6	14/11/2018	Recent	4	4	2WE	2WE	3.8
MM1	15/11/2018	Mid	6	6	2WE	2WE	7.2
O1	15/01/2019	Mid	9	9	2WE	2WE	11.9
TP2	20/11/2018	Mid	8	8	2WE	2WE	5.2
TP1	19/11/2018	Mid-late	12	12	2WE	2WE	6.7
R2	22/11/2018	Native	N/A	N/A	NF	NF	N/A
T1	12/11/2018	Pasture	0	0	NF	NF	0
BC2	10/12/2018	Recent	5	5	NF	6WPB	>20.0
AA1	27/11/2018	Mid-late	13	13	NF	8W+1E	>20.0
BC1	14/12/2018	Mid-late	11	11	5W3E	5W3+1E	14.9
OM1	18/01/2019	Mid-late	14	14	6WPB+1E	6WPB+1E	>20.0
TA1	28/11/2018	Mid-late	12	12	6WPB	NF	16.8
TM1	21/01/2019	Old	16	16	6WPB+1E	6WPB+1E	7.7
W1	13/12/2018	Old	23	N/A	NF	NF	>20.0
W2	7/12/2018	Native	N/A	N/A	NF	NF	N/A
K3	7/11/2018	Pasture	0	0	NF	NF	0
TU2	16/01/2019	Recent	5	5	5W+2E	5W2+1E	4.4
TU1	17/01/2019	Recent	1	10	8WPB	8WPB	>20.0
K1	31/11/2018	Old	18	18	6WPB	6WPB	>20.0
K2	12/12/2018	Old	18	18	6WPB	6WPB	>20.0
WK1	11/12/2018	Native	N/A	N/A	NF	NF	N/A

Chapter 3

Influence of upstream catchment and riparian characteristics on instream habitat

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3.1 Introduction

Increased habitat heterogeneity is regarded as a key factor that positively influences restoration or enhancement of biodiversity. For stream ecosystems, increased habitat heterogeneity can provide more refugia, broader niche area and greater variation and quantity of trophic resources (Palmer *et al.*, 2010). Accordingly, physical habitat is a primary factor influencing composition and structure of stream communities, however, its influence occurs across a range of spatial and temporal scales (Lammert & Allan, 1999).

Frissell *et al.* (1986) described stream systems as hierarchically-organised, spatially-nested systems with each level forming the environment for the levels below. The hierarchy of levels cascades from catchment to segment, reach, pool/riffle system and finally to microhabitat, with the larger scale features constraining the characteristics of the smaller units. Winterbourn (1981) described the environmental factors influencing lotic invertebrate distribution in New Zealand streams as a hierarchy of: i) large-scale catchment factors including location, altitude, geology, vegetation and water chemistry; ii) medium-scale factors such as stream size, flow regime, gradient, bed stability, food supply and riparian vegetation; and iii) small-scale factors such as velocity, substrate size and local physicochemical parameters such as temperature and dissolved oxygen availability.

Large catchment-scale influences on stream ecosystems are dominated by hydrology, a function of climate, catchment size, underlying geology and topography. Hydrology influences stream flow volume, flow variability and water velocity which interact with gradient and geology to affect channel shape and benthic substrate composition (Jowett *et al.*, 1996; Allan & Castillo, 2007).

Catchment vegetation, particularly forest cover, influences hydrology through canopy rainfall interception and evapotranspiration and by increasing absorption capacity of soils, thereby reducing surface run-off and flooding potential (Ministry for the Environment, 2001). Suspended sediment loads are influenced by catchment geology and hydrology, upstream land-use, land management and vegetation cover (Lintern *et al.*, 2018). The cumulative influences of these factors determine the size, variability and distribution of habitats such as riffle, run, pool sequences and substrate composition which in turn drive the function and structure of ecosystems (Death *et al.*, 2015).

Riparian vegetation can influence instream habitat at both catchment and reach-scales by: i) provision of shade and subsequent reduction in water temperature and primary productivity (Davies-Colley & Quinn, 1998; Rutherford *et al.*, 1999; Quinn *et al.*, 2007); ii) increased inputs of woody debris and coarse particulate organic matter (CPOM); iii) increased habitat complexity by provision of root complexes, undercut banks, and overhanging vegetation (Parkyn *et al.*, 2009; Quinn *et al.*, 2009); iv) bank stabilisation, mitigation of overland flow, reduction in sediment and increased nutrient attenuation (Wilcock *et al.*, 2013); and v) provision of connectivity and corridors across aquatic and terrestrial landscapes for dispersal and re-colonisation (Ministry for the Environment, 2001).

In New Zealand, widespread removal of riparian vegetation has accompanied pastoral development, influencing the biological and physicochemical nature of streams (Townsend *et al.*, 1997). Concerns regarding the impact of pastoral land-use on streams in New Zealand have primarily focused on degradation of water quality (Howard-Williams *et al.*, 2010), with effects on instream habitat often the forgotten dimension of stream ecosystem health and rarely receiving mention. Declining stream health has fuelled both regulatory and voluntary changes in land management practices to ameliorate agricultural impacts on stream ecosystems. One of the most used tools for mitigating pastoral land-use effects is the reinstatement of riparian vegetation, generally through fencing followed by revegetation of riparian zones (Greenwood *et al.*, 2012; McKergow *et al.*, 2016).

A range of factors require consideration in regard to the effectiveness of riparian rehabilitation at mitigating pastoral land-use effects. At a catchment-scale, the effectiveness of riparian management depends on several upstream constraints such as: i) the distance the managed reach is from forested headwaters; ii) the spatial arrangement of planted reaches and their degree of connectivity; iii) location in the catchment; iv) stream size; and v) historic and contemporary land-use in upstream areas (Parkyn *et al.*, 2003; Dodds & Oakes, 2008; Greenwood *et al.*, 2012).

At a reach-scale, the impact of riparian management on instream habitat can depend on stream size, width of the managed area, species planted, planting density and length of planted reaches (Parkyn *et al.*, 2000; Parkyn *et al.*, 2003; Davies-Colley *et al.*, 2009). Holmes *et al.* (2016) suggested at least 300 m of upstream fenced riparian area were required to provide a positive instream habitat response in the form of reduced fine sediment deposition, while modelling results from Rutherford *et al.* (1999) suggested at least 1-5 km of first order stream or 10-20 km of fifth order streams need to be planted to achieve 75% shade and reduce daily maximum water temperature by 5°C. Greenwood *et al.* (2012) also found limited responses to riparian restoration in degraded Canterbury streams with long land-use legacies, suggesting that riparian management may not improve local instream habitat but may reduce downstream effects.

Responses of streams to riparian fencing and planting occur over a variety of timeframes. In small (1-2 m wide), pasture streams at Whatawhata, reductions in stream water temperature were observed as early as six years after fencing and native planting (Quinn *et al.*, 2009), while pine afforestation created shade close to levels found in native forest streams within 15 years of planting (Quinn *et al.*, 1997). At the same site, macroinvertebrate community indices increased within eight years in response to restoration of riparian vegetation (Jowett *et al.*, 2009).

The relative influence of catchment processes versus reach-scale restoration of riparian habitat on instream habitat is poorly understood. Quantifying the relationships between essential habitat characteristics and target species, along with the key influences on these habitats are critical to maintain healthy populations of target species (Death *et al.*, 2015).

The inherent variability of stream habitats induced by catchment-scale and reach-scale influences can make measuring the success of riparian management problematic due to the influence of underlying wider catchment parameters as well as land management, variability of riparian management and other associated land-use practices (Belsky *et al.*, 1999). Accordingly, the aim of this chapter is to determine factors influencing kōura habitat in 26 small Waikato pastoral streams within managed riparian areas of different-aged plantings to resolve the relative importance of catchment-scale versus reach-scale influences on the outcomes of riparian management.

Specific objectives for this chapter are:

- Establish the influences of upstream catchment factors on reach-scale instream habitat characteristics. The upstream catchment factors investigated were catchment size, geology, land-cover type (vegetation), stream gradient, distance upstream to native forest and proportion of upstream area with riparian vegetation.
- Determine the influence of reach-scale riparian planting age on instream habitat variables of relevance to kōura, including shade, and abundance of root complexes, woody debris, coarse particulate organic matter (CPOM), undercut banks, overhanging vegetation and macrophyte cover.

Based on previous findings from international sources, as well as from New Zealand, the following hypotheses were tested in this chapter:

1) Upstream catchment characteristics will significantly influence reach-scale habitat at survey sites, with upstream catchment size, land-cover and geology exerting the strongest influence;
2) Older riparian plantings (≥11 years) will provide more shade and important instream habitat associated with kōura abundance such as root complexes, woody debris, CPOM, undercut banks and overhanging vegetation and less macrophytes, than those in pasture or ≤ 5 years old;
3) Older riparian plantings (16+ years old) will closely resemble native forest sites in terms of shade and important kōura habitat variables, noteably root complexes, woody debris, CPOM, undercut banks and overhanging vegetation.

3.2 Methods

3.2.1 Upstream catchment characteristics

Geographic Information Systems (GIS) analysis was used to calculate a range of upstream catchment and sample reach characteristics for each study site. All analysis was undertaken using QGIS v2.18, Open Source software, Switzerland.

Upstream catchment boundaries and size

Upstream catchment boundaries were delineated based on the Land Information New Zealand 8 m resolution Digital Elevation Model (DEM)[3]. The GRASS 'r.watershed.water' basin analysis algorithm was used with the DEM to determine flow direction and the number of upslope 20 m x 20 m cells flowing into each cell. Results provided input data to run the catchment area analysis in the GRASS 'r.water.outlet' algorithm in combination with the spatial co-ordinates for the upstream end of each sample site. The resulting output was a delineated upstream catchment boundary for each sample reach.

Catchment boundaries were checked for errors against NZTM 2000 topographic map layers and 2012 and 2017 Waikato Regional Council Aerial Photo Survey (WRAPS)[4] layers produced by Waikato Regional Council (WRC). Sites BC1, BC2, OM1 and the downstream end of TM1 (see Figure 2.1) only had 2012 photography available at the time of analysis. All other sites had 2017 or 2018 images which are of a greater resolution than the 2012 images. The QGIS field calculator was used for calculating catchment area (ha) within the delineated catchment boundaries.

Distance of upstream stream network

To estimate stream length above each sample reach, both the NZTM 2000 topographical maps[5] and the WRC River Environment Classification[6] GIS layers were compared to catchment aerial photos for their accuracy. Both lacked the detail considered appropriate for this type of analysis so a process of hand digitising the stream network from each site was undertaken.

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[3] Source: Land Information New Zealand, creative commons.
[4] Source: Waikato Regional Council, creative commons.
[5] Source: Land Information New Zealand, creative commons.
[6] Source: Waikato Regional Council, Data Use Agreement DOC#12310105.

To ensure consistency of the extent of the stream network included in the analysis, a minimum upstream catchment size of >20 ha was used to represent the upstream extent of perennial stream which could provide habitat for kōura. In catchments <20 ha it is likely that many channels will be intermittent or ephemeral so this was used as a threshold for delineating perennial stream length, although this catchment size has been shown to vary in differing geologies (Storey & Wadhwa, 2009). This minimum upstream catchment size is also the channel extent used for the REC and therefore provides consistency with other methods applied in the New Zealand setting.

The ‘r.water.outlet’ output layer was set to show cells receiving input >20 ha and was used in conjunction with NZTM 2000 and WRC WRAPS (2012 and 2017/18) aerial photography to hand digitise the stream networks. Where the stream was visible on aerial photos, it was digitised to fit the channel as closely as possible given photograph resolution. Where the stream was obscured by dense vegetation, the stream channels were matched to the ‘r.water.outlet’ layer. The QGIS field calculator was used to calculate upstream channel length from analysis on the digitised stream network.

Extent of upstream riparian buffer and vegetation

The digitised stream GIS layer was broken into a series of 50-m segments using the GRASS ‘v.split.length’ tool. Using the ‘v.buffer.distance’ tool, a buffer of ten metres either side of each stream segment was applied based on recommendations by Parkyn *et al.* (2000) that a riparian buffer ≥10 m should be sufficient to achieve a range of aquatic functions in small streams. Thus, the polygons produced from this process were 20-m wide and 50-m in length along the stream line, equivalent to approximately 1,000 m^2 in size. Due to the natural curvature of stream channels, the area of segments varied.

The buffer segment layer was joined to the WRC Biodiversity Vegetation - BIOVEG (2012)[7] GIS layer. The BIOVEG layer contains parcels of terrestrial vegetation >0.5 ha mapped from the WRC WRAPS 2012 aerial photography and is based on a simplified version of the Land Cover Database classifications for the

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[7] Source: Waikato Regional Council, creative commons.

Waikato Region. Segment polygons that intersected the BIOVEG layer were automatically given a value of 100% vegetation cover. The BIOVEG layer was utilised to reduce the need to assess vegetation cover in many polygons which were located in contiguous native forest.

All polygons (including those that intersected the BIOVEG layer) were then visually checked or assessed against WRC WRAPS (2012 and 2017/18) aerial photographs to estimate percentage cover of vegetation using the scale outlined in Table 3.1. Only vegetation considered ‘woody’ such as shrubs, trees or native and exotic forest identified in aerial photos was included in the cover assessment. Some polygons that intersected the BIOVEG layer contained pasture so the automatic allocation of 100% cover was adjusted if required. For the purposes of the assessment, no distinction was made between exotic and native vegetation types, and nor was any interpretation made as to whether the buffer was likely to be fenced to exclude stock and/or had undergone riparian planting.

Table 3.1: Woody and shrub vegetation cover categories for 50 x 20 m riparian buffer segments of stream networks

Assessed vegetation cover (% of each stream buffer polygon)	**Percent shrub and tree cover value allocated**
0-5	0
6-15	10
16-25	20
26-35	30
36-45	40
46-55	50
56-65	60
66-75	70
76-85	80
86-95	90
96-100	100

The QGIS field calculator was used to calculate the size of each stream buffer polygon (ha). Hectare values were used to calculate the extent of woody vegetation in each polygon based on the sum of the polygon area and the estimated percentage cover. Total woody vegetation cover for each upstream catchment buffer was then calculated by summing the values for the individual polygons.

Upstream distance to native forest

A second GIS layer was created using the hand digitised stream network such that all streams were clipped to where contiguous native forest started. For streams which had more than one tributary flowing from native forest, the shortest tributary was used in this analysis. Distance to native forest from the upstream end of each sample reach was then calculated using the QGIS field calculator. An ordinal scale was developed to represent distance categories for analysis purposes as some sites had no native forest in the catchment and native sites had zero distance making it difficult to undertake robust analysis using distance values. The categories used were: 1 = within native forest, 2 = <1 km to native forest, 3 = 1-3 km, 4 = >3 km, 5 = pasture/no native present.

Upstream and sample reach channel gradients

The DEM was used to determine the highest elevation (m above sea level (asl)) of each catchment draining into the delineated stream network. Upstream channel gradient was calculated from this point to the upstream end of the sample reach, and the upper and lower elevations of the sample reach were used to calculate sample reach gradient based on the digitised stream length.

Upstream catchment geology and land-cover

The New Zealand Land Resource Inventory (NZLRI) was used to establish dominant geology (top rock) for the study sites and the New Zealand Land Cover Database (LCDB) Version 4.1 was used to establish the main types of vegetation within each study catchment[8]. Data categories were amalgamated for: i) indigenous forest and indigenous hardwood and manuka/kanuka categories (indigenous); ii) high and low production grassland (grass); and iii) all other land-

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[8] NZLRI and LCDB data reproduced with the permission of Landcare New Zealand Limited

cover categories including exotic forest, harvested forest, settlement, deciduous hardwoods and lakes (“other” vegetation). The gorse and broom category was not amalgamated and is reported as scrub.

Limitations for upstream catchment categorisation

Although efforts were made to ensure a consistent and robust process during the GIS analysis of upstream catchment and stream channel characteristics, there remain some limitations in the data. Specifically, these are: i) the use of >20 ha as the point of initiation for catchments with perennial stream flow likely underestimated the stream network length; ii) resolution of aerial photos for 2012 is limited so delineation of the stream channel and quantification of woody buffer vegetation was difficult; and iii) although resolution of 2017 aerial photos was higher, areas of woody vegetation and the location of the stream channel had to be estimated in some cases. Accordingly, the data produced from this GIS process provide estimated, (but comparable) upstream catchment-scale characteristics among sites for the purposes of subsequent analyses.

3.2.2 Riparian characteristics

Fencing type for each side of the 50 m, riparian-planted sample reaches was recorded. The age of fencing and the age of plantings were ascertained through WRC GIS information and through discussions with landowners.

For riparian vegetation categorisation, three plots at 0-10, 20-30 and 40-50 m from the downstream end of the reach were utilised on each side of the stream (i.e., $n = 6$) to provide an average indication of the extent and type of riparian vegetation and cover present within the sample reach. The width of each riparian vegetation plot was equivalent to the adjacent stream channel wetted width, with a minimum plot width of 2 m such that plot areas were ≥ 20 m^2 and scaled according to stream width.

Wetted and channel width, along with slope of the stream banks and stream aspect were measured at the mid-point of each of the three sub-reaches used for riparian vegetation characterisation.

To determine the area of wood within the riparian plots, diameter at breast height (DBH) was measured for all stems ≥1.35 m height and ≥2.5 cm diameter using a DBH measuring tape. All stems on every tree which met these criteria were measured, and the species of tree or shrub identified. Standing dead wood meeting these criteria was also measured and recorded. Basal area was calculated using the formula: $\mathrm{BA}(\mathrm{m}^2) = \frac{\mathrm{pi}(3.14)\times \mathrm{d}^2}{4} \times 10{,}000$ (Kershaw *et al.*, 2017) where d is diameter, and then converted to basal area m^2 per 20 m^2 area sampled.

Woody plants or tree ferns >1.35 m tall with <2.5 cm DBH were classified as saplings and counted. Tree ferns were included where they had a natural frond height >1.35 m and stem length <1.35 m. Flax was counted if plants had leaf height >1.35 m.

To quantify the lower tiers of riparian and ground vegetation, the following categories were estimated as percent cover within the sample area: i) bare ground; ii) mosses and lichen; iii) detritus and leaf litter; iv) grass and pasture weeds <0.3 m height; v) rank grass, pasture weeds or other vegetation >0.3 m in height; and vi) shrubs and sedges/grasses >0.3 m and less than 1.35 m (shrubs). Values within plots for each bank totalled 100%. If present, the proportions of trees, shrubs and sedges/grasses that were native or exotic were recorded for each sample area. Values did not necessarily total 100% as some sample areas had no vegetation within these categories.

Riparian measurements were collected either prior or after kōura and fish sampling as described in Chapters 4 and 5, respectively.

3.2.3 Instream habitat characteristics

All sample reaches were 50-m in length and were divided into five 10-m sub-reaches.

Quantification of instream habitat characteristics

At the mid-point of each of the five sub-reaches channel and wetted width and thalweg and mid-channel depths were measured.

Mid-channel shade was measured at the middle of each sub-reach using a spherical densitometer (Forestry Suppliers Limited, 2008). Measurements were made by estimating whether overhead cover >50% occurred for each of the four points in each 24 grid squares (i.e. 96 points) on the densiometer. Shade was assessed facing upstream and downstream as well as toward the true left and right banks, averaged for each sub-reach and multiplied by 1.04 to provide percentage shade as canopy cover, then averaged across the five sub-reaches to obtain a reach-scale estimate.

Within the wetted channel of each sub-reach, ten substrate elements spaced across the mid-point of each transect were measured across the b-axis on the Uggden/Wentworth scale (Clapcott, 2015). In addition, total bed area covered (m^2) was measured for CPOM (larger than 1 mm and less than 10 cm diameter) and total cover of large woody debris >10 cm in diameter and >20 cm along longest axis. Lengths (m) of stable undercuts present on each bank and total area (m^2) of root complexes present along each bank or in the channel were also measured. The percentage length of each stream bank with overhanging vegetation <30 cm above the water surface was estimated along with percentages of run, riffle, pool, backwater or other types of flow habitats, and percent of bed area covered by macrophytes.

Instream habitat measurements which may have been compromised by electric-fishing disturbance were collected prior to kōura sampling and the remaining structural measurements collected at the conclusion of electric-fishing.

3.2.4 Statistical analyses

Upstream catchment and riparian characteristics

Principal Component Analysis (PCA, PRIMER 7.0.13) was used to explore relationships amongst sites and site clusters using upstream catchment and riparian variables. With the exception of ordinal scales, data were transformed using log(x+1) for values and square-root for percentages and then normalised. Vector plots used correlation loadings of >0.3.

Factors influencing instream habitat

PCA was used to explore relationships between instream habitat variables amongst sites and site spatial clusters. With the exception of ordinal scales, data were transformed using log(x+1) for values and square-root for percentages and then normalised. The two-dimensional PCA plot was overlain with riparian and catchment vectors using correlation loadings of >0.3.

Step-wise distance-based linear models (DistLM, PRIMER 7.0.13) were used to test which combination of upstream catchment and riparian variables best explained the dissimilarities in instream habitat between sites and created the most parsimonious model. All sites were entered into the analysis (i.e., native forest, riparian and pasture). The response variable in these models was the dissimilarity matrix based on the Euclidean distances between transformed and normalised instream habitat variables. Draftsman plots and correlation matrices were used to identify strongly correlated (r >0.8) variables within the upstream catchment and riparian datasets. Where strong correlations existed, one variable was removed prior to further analysis. Data were also checked for right skewness which can influence DistLM results. None was found.

Using the instream habitat resemblance matrix, marginal tests and manual selection tests were performed individually on transformed, normalised riparian and upstream catchment variables. Adjusted R^2 was used as a criterion for variable selection as it takes into account the number of parameters and excludes predictor variables if they add no more to the explained sum of squares. Variables which contributed <3% explanatory power in manual and forward selection models were also excluded. The results of the manual selection process were then compared to combinations returned using the forward selection and best-fit process.

The best-fit process produces only total adjusted R^2 and cumulative R^2 values representing all the variables contained in the model rather than for individual variables making it difficult to establish if individual variables contribute >3% explanatory power to a particular model. To minimise differences when comparing models, the number of variables in the best-fit models were compared to manual selection results. As a guide, best-fit models with the same number of variables as manual selection models or with approximately half the variables included in the original analysis are reported on. Comparisons between manual, forward selection and best-fit processes, in conjunction with logic based on knowledge of streams and the interaction of habitat variables, enabled selection of models for upstream catchment and riparian variables that were intuitive, parsimonious and had relatively good predictive power.

Results from the upstream catchment and riparian models were then reanalysed in various combinations using the manual selection process, then combined and rerun using forward selection and best-fit to establish the most logical and parsimonious model that accounted for the most variation in instream habitat between sites.

Effects of riparian planting age on instream habitat

Two-way PERMANOVA (PRIMER 7.0.13) was used to test for overall differences in instream habitat using riparian age group and site spatial cluster as factors. Analysis used the dissimilarity matrix based on the Euclidean distances between transformed and normalised instream habitat variables.

Non-parametric Kruskal-Wallis test (STATISTICA v13.3) was used to determine ranked differences in seven instream habitat variables associated with riparian vegetation among different-age classes of riparian plantings. Riparian sites were classified into five-year age blocks of 1 = pasture (i.e., zero riparian age), 2 = 1-5 years or recent riparian age, 3 = 6-10 years or mid riparian age, 4 = 11-15 years or mid-late riparian age, and 5 = 16+ years or old riparian. Native sites were excluded from this analysis to test the null hypothesis that there would be no difference in instream habitat variables between riparian-planting age groups at sites formerly or currently grazed.

Non-parametric Mann-Whitney U tests (STATISTICA v13.3) were performed on two combinations of different riparian age groups for the seven instream habitat variables associated with riparian vegetation to establish whether sites with older riparian plantings are associated with instream habitat characteristics similar to native forest sites. The age classes compared were 11-15 years (mid-late) versus native forest sites, and 16+ years (old) versus native forest sites.

3.3 Results

3.3.1 Upstream catchment and riparian characteristics

A summary of upstream catchment and riparian variables across different site types is shown in Table 3.2 and Table 3.3, respectively.

For the upstream catchment and riparian data for each individual site refer to Appendix 1 and Appendix 2.

As part of GIS catchment analysis, the shortest distance upstream from the top of each sample reach to contiguous native forest cover was calculated for all sites (see Section 3.2.1). Data are not presented as the results required classification of each site into an ordinal scale to allow analysis.

Sites BC1, O1, OM1, TP2, TP3 and K2 (see Figure 2.1) had no contiguous native forest in the upstream network identified using the GIS layer delineated by digitising catchments >20 ha. Site K1 was located in remnant native forest with riparian planting, however, the upstream catchment was <20 ha so no upstream catchment was digitised. All native sites had their entire upstream catchment in contiguous native forest. Of the remaining 16 sites (three pasture and 13 riparian), the mean upstream stream length was 2,422 m ± SD 1662. Distance varied from a minimum of 322 m to a maximum of 5,340 m.

Table 3.2: Mean ± standard deviation (SD) and minimum and maximum values of upstream catchment variables by category for study sites. Pasture n = 3, riparian n = 20, native n = 3.

Variable	**Pasture**				**Riparian**				**Native**			
	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Catchment area (ha)	311.0	± 262.8	80.0	597.0	217.8	± 202.6	18.0	794.0	224.0	± 161.9	37.0	318.0
Riparian buffer in woody vegetation (%)	69.8	± 26.7	39.0	86.3	58.3	± 26.1	8.6	97.2	97.0	± 5.1	91.0	100
Grass vegetation (%)	29.0	± 14.0	15.0	43.0	56.2	± 23.8	17.0	100	6.3	± 10.9	0	19.0
Indigenous vegetation (%)	67.6	± 14.3	57.0	84.0	33.2	± 25.5	0	83.0	93.0	± 11.6	80.0	100
Scrub vegetation (%)	0.6	± 1.1	0	2.0	2.8	± 9.4	0	42.0	0	0	0	0
Other vegetation (%)	2.6	± 3.7	0	7.0	7.7	± 18.7	0	64.0	0.6	± 0.5	0	1.0
Gradient (m/m)	0.1	± 0.1	0	0.1	0.1	± 0	0	0.1	0.1	± 0.1	0.02	0.1
Ashes older than Taupō eruption (%)	7.3	± 6.5	0	12.5	15.4	± 23.9	0	93.1	0.7	± 1.2	0	2.2
Volcanic rock (%)	56.5	± 49.0	0	87.4	59.7	± 41.8	0	100	65.8	± 57.0	0	100
Greywacke (%)	33.3	± 57.7	0	100	19.2	± 36.9	0	100	33.3	± 57.7	0	100
Sandstone (%)	0	0	0	0	3.5	± 10.9	0	41.8	0	0	0	0
Mudstone (%)	0	0	0	0	1.1	± 3.4	0	14.2	0	0	0	0
“Other” geology (%)	2.7	± 4.7	0	8.2	0.9	± 2.8	0	9.7	0	± 0.1	0	0.2

Table 3.3: Mean ± standard deviation (SD) and minimum and maximum values of riparian variables by category for study sites. Pasture n = 3, riparian n = 20, native n = 3.

Variable	**Pasture**				**Riparian**				**Native**			
	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Average stream bank slope (°)	19.3	± 16.5	8.5	38.3	20.1	± 7.8	6.3	35.2	18.9	± 8.1	12.3	28.0
Average buffer width (m)	0	0	0	0	12.5	± 6.5	3.8	20.0	N/A	N/A	N/A	N/A
Number of saplings (per m^2)	0.1	± 0.1	0	0.1	0.2	± 0.2	0	0.7	0.3	± 0.2	0.1	0.4
Basal area (m^2 per 20 m^2)	0.1	± 0.1	0	0.1	0.1	± 0.1	0	0.2	0.1	± 0.1	0.1	0.2
Bare ground (%)	6.9	± 4.8	3.3	12.5	3.0	± 3.6	0	10.0	5.6	± 6.9	0	13.3
Mosses/liverworts (%)	0	0	0	0	2.4	± 5.1	0	21.7	14.4	± 13.5	6.7	30.0
Leaf litter (%)	0.3	± 0.5	0	0.8	8.6	± 13.0	0	55.8	29.2	± 11.2	20.0	41.7
Vegetation ≤0.3 m in height (%)	68.9	± 34.0	30.0	93.0	0.9	± 2.7	0	11.7	1.1	± 1.9	0	3.3
Vegetation >0.3 m in height (%)	20.3	± 35.1	0	60.8	49.5	± 26.6	0	87.5	8.9	± 8.4	0	16.7
Shrubs (%)	3.6	± 1.7	1.7	5.0	35.6	± 16.2	10.0	70.0	40.8	± 5.1	35.0	44.2
Shrubs and saplings (native) in buffer (%)	47.2	± 12.7	33.3	58.3	89.0	± 15.8	50	100	99.4	± 1.0	98.3	100
Shrubs and saplings (exotic) in buffer (%)	2.8	± 4.8	0	8.3	9.4	± 12.9	0	50.0	0.6	± 1.0	0	1.7

PCA was carried out to determine which upstream catchment variables best explained variation between sites. The first two axes accounted for 51.2% of the variation, while four axes account for a total of 72.2% (Table 3.4).

Table 3.4: Eigenvalues and percent variation for each principal component (PC) axis for upstream catchment variables across 26 study sites

PC axis	Eigenvalue	% Variation	Cumulative % Variation
1	4.1	29.7	29.7
2	3.0	21.5	51.2
3	1.7	12.0	63.2
4	1.3	9.0	72.2

The PCA plot with upstream catchment variable overlay (Figure 3.1), indicated the majority of Whaingaroa and Pirongia sites clustered towards the left of the plot regardless of riparian planting age. These sites are associated with volcanic rock and pre-Taupō ash geology and higher upstream gradients, capturing the steeper nature of these streams compared to the Whatawhata sites. Most Pirongia sites and two Whaingaroa sites (native and 23 years old) showed a stronger association with percentage indigenous vegetation indicative of their proximity to or inclusion in native forest found on Mount Pirongia and Mount Karioi. Geology differs for Whatawhata sites which were grouped towards the right of the plot and associated with greywacke, sandstone and mudstone top rock and extent of upstream buffer vegetation. One Pirongia site (KA1) was more closely associated with the Whatawhata sites, reflecting its geographical location mid-way between Pirongia and Whatawhata and the subsequent transition in geology.

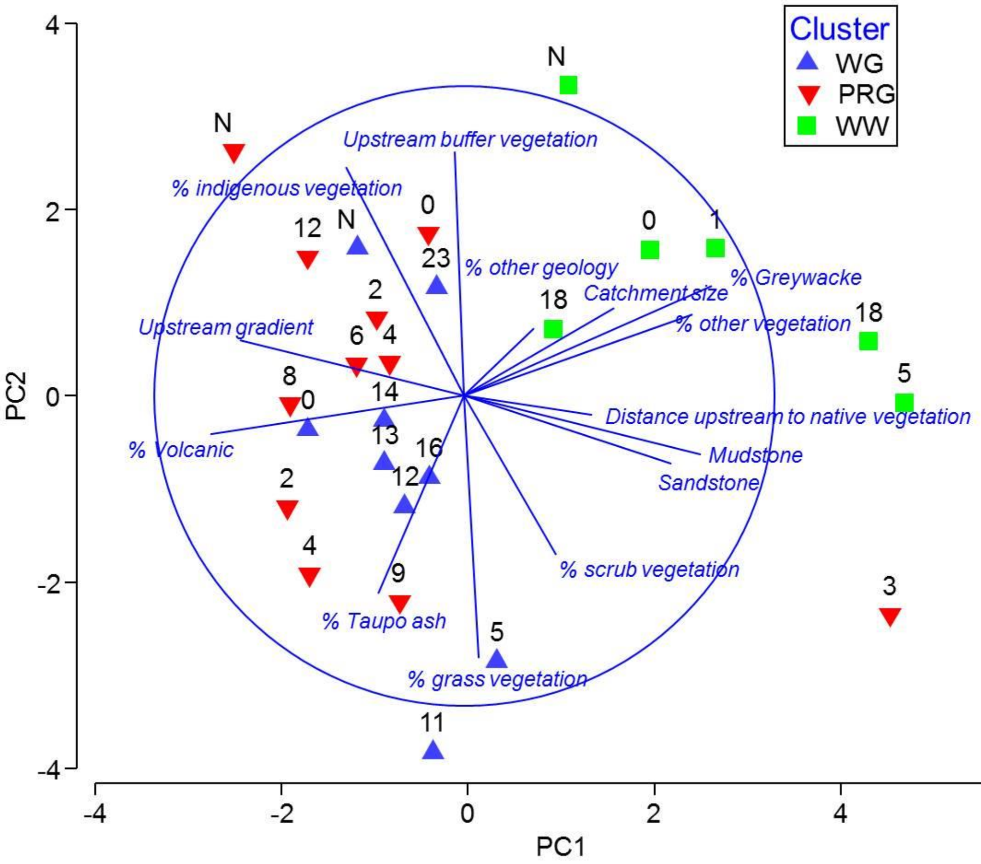


Figure 3.1: PCA plot based on upstream catchment variables of all study sites by spatial cluster and riparian planting age along principal component (PC) axes 1 and 2, showing relationships with upstream catchment variables with *r* >0.3. Clusters are: WG = Whaingaroa, PRG = Pirongia, WW = Whatawhata. Numbers refer to riparian planting age (years). Native study sites are marked N and pasture sites as “0”.

Examination of the PC axes and factor loadings reveal PC axis one primarily represented the key classes of geology for volcanic rock, greywacke, mudstone and sandstone followed by other vegetation and upstream gradient (Table 3.5). PC axis two associated primarily with upstream catchment vegetation such as percentage land-cover in grass and indigenous vegetation, percentage woody buffer vegetation and also pre-Taupō ash geology. PC axis three associated most strongly with percentage “other” geology, upstream catchment size and distance upstream to native forest, while PC axis four also represented percentage “other” geology, distance to upstream native forest and land-cover in scrub vegetation.

Table 3.5: Factor loadings of each upstream catchment variable for the first four principal component (PC) axes. Values >0.3 and <-0.3 are shown in bold.

Variable	**PC1**	**PC2**	**PC3**	**PC4**
Catchment area (ha)	0.237	0.164	**0.457**	0.191
Riparian buffer in woody vegetation (%)	-0.015	**0.455**	-0.217	0.162
Grass vegetation (%)	0.023	**-0.488**	-0.142	0.236
Indigenous vegetation (%)	-0.186	**0.426**	0.238	-0.003
Scrub vegetation (%)	0.145	-0.295	0.234	**-0.431**
“Other” vegetation (%)	**0.361**	0.151	-0.112	-0.046
Gradient (m/m)	**-0.354**	0.104	0.168	-0.163
Ashes older than Taupō eruption (%)	-0.135	**-0.367**	-0.007	0.259
Volcanic rock (%)	**-0.401**	-0.071	**0.307**	-0.021
Greywacke (%)	**0.392**	0.204	-0.299	-0.133
Sandstone (%)	**0.327**	-0.126	0.229	0.287
Mudstone (%)	**0.374**	-0.108	0.130	0.070
“Other” geology (%)	0.110	0.126	**0.453**	**0.418**
Distance to native forest (ordinal scale)	0.202	-0.035	**0.337**	**-0.565**

Table 3.6: Eigenvalues and percent variation for each principal component (PC) axis for riparian variables across 26 study sites.

PC axis	**Eigenvalue**	**% Variation**	**Cumulative % Variation**
1	3.8	38.2	38.2
2	2.4	24.4	62.5
3	1.2	11.8	74.3
4	0.8	8.1	82.5

The PCA of riparian variables indicated the first two axes account for 62.5% of the variation between sites while the cumulative variation of the first four axes explained 82.5% (Table 3.6).

The riparian variable overlay indicated no spatial clustering but some grouping based on riparian age (Figure 3.2). The majority of younger sites from all clusters were positively associated with percentage vegetation >0.3 m high and were grouped towards the top right of the PCA plot. Two pasture sites (indicated by “0” in Figure 3.2) were strongly associated with percentage vegetation ≤0.3 m high while the Whatawhata pasture site was closer to the percentage vegetation >0.3 m high vector consistent with less intensive grazing at this site due to the steep and incised nature of the stream banks. Most of the older riparian and native sites were grouped towards the top left of the PCA plot associated with wider riparian buffers (buffer width), percentage shrub cover, higher numbers of saplings and greater basal area.

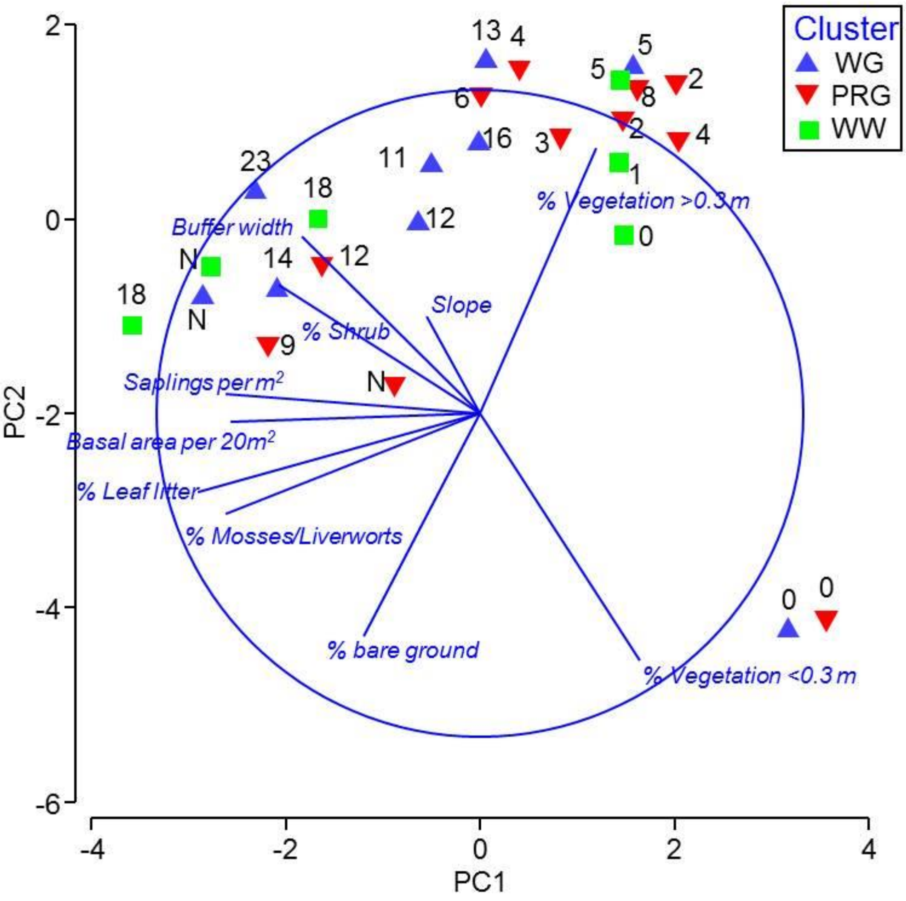


Figure 3.2: PCA plot based on riparian variables of all study sites by spatial cluster and riparian planting age along principal component (PC) axes 1 and 2, showing relationships with riparian variables with *r* >0.3. Clusters are: WG = Whaingaroa, PRG = Pirongia, WW = Whatawhata. Numbers refer to riparian planting age (years). Native study sites are marked N and pasture sites as “0”.

Examination of the PC axes factor loadings (Table 3.7) indicate PC axis one primarily represented variables associated with the presence of woody trees and shrubs such as leaf litter, saplings, mosses and liverworts, basal area and shrubs. PC axis two associated more strongly with variables related to lower growing vegetation (>0.3 m and ≤ 0.3 m), bare ground and average buffer width. Average stream bank slope and shrubs were mainly represented by PC axis three, with PC axis four strongly associated with stream bank slope followed by average buffer width and bare ground.

Table 3.7: Factor loadings of each riparian variable for the first four principal component (PC) axes. Values >0.3 and <-0.3 are shown in bold.

Variable	**PC1**	**PC2**	**PC3**	**PC4**
Average stream bank slope (°)	-0.084	0.191	**-0.717**	**0.534**
Average buffer width (m)	-0.283	**0.351**	-0.035	**-0.484**
Number of saplings (per m^2)	**-0.403**	0.040	-0.185	-0.246
Basal area (m^2 per 20 m^2)	**-0.395**	-0.017	-0.225	-0.205
Bare ground (%)	-0.185	**-0.441**	-0.166	**-0.379**
Mosses/liverworts (%)	**-0.403**	-0.200	0.235	0.250
Leaf litter (%)	**-0.446**	-0.155	-0.149	0.204
Vegetation ≤0.3 m in height (%)	0.252	**-0.489**	-0.200	-0.119
Vegetation >0.3 m in height (%)	0.185	**0.526**	-0.211	-0.246
Shrubs (%)	**-0.319**	0.254	**0.459**	0.235

To examine the patchiness of the riparian vegetation at study sites, coefficients of variation (SD/mean) were calculated for basal area and sapling values among riparian sample plots (n = 6 per site) within each riparian planting age group (Table 3.8). For basal area (m^2 per 20 m^2), variability was highest within recent plantings (1.25 ± SD 0.38) followed by mid-late plantings (0.75 ± SD 0.59). The mid (0.43 ± SD 0.16) and old (0.49 ± SD 0.37) age group categories had similar variation while both pasture (0.28 ± SD 0.33) and native (0.37 ± SD 0.23) sites had low variability.

For sapling counts (no. per m^2), mid-aged riparian plantings showed the highest variability (0.88 ± SD 0.74), followed by native (0.78 ± SD0.45), recent (0.76 ± SD 0.31), old (0.56 ±SD 0.45) and mid-late (0.54 ± SD 0.43). Pasture sites showed the least variation (0.53 ± SD 0.31).

Table 3.8: Mean coefficients of variation (CV) ± standard deviation (SD) between sites for each age-group category for basal area (m^2 per 20 m^2) and number of saplings (per m^2).

Age group	**Basal area (m^2 per 20 m^2)**		**Number of saplings (per m^2)**	
	CV	SD	CV	SD
Pasture (no planting)	0.28	± 0.33	0.53	± 0.31
Recent (1-5 years)	1.25	± 0.38	0.76	± 0.65
Mid (6-10 years)	0.43	± 0.16	0.88	± 0.74
Mid-late (11-16 years)	0.75	± 0.59	0.54	± 0.43
Old (16+ years)	0.49	± 0.37	0.56	± 0.45
Native	0.37	± 0.23	0.78	± 0.45

To gauge vegetation composition of riparian sample areas, the proportion of native versus exotic vegetation was determined. When trees, shrubs, saplings and sedges/grasses >0.3 m high were present at pasture sites, 47.2% (± SD 12.72) was native while <3% (± SD 4.81) was exotic. All riparian sites had some type of vegetation present with the majority being native (88.9% ± SD 15.77). As expected, native forest sites were dominated by native vegetation (99.4% ± SD 0.96).

3.3.2 Instream habitat characteristics

A summary of instream habitat variables across different site types is shown in Table 3.9. For instream habitat data for each individual site refer to Appendix 3.

Table 3.9: Mean ± standard deviation (SD) and minimum and maximum values of instream habitat variables by treatment for study sites. Pasture n = 3, riparian n = 20, native n = 3.

Variable	**Pasture**				**Riparian**				**Native**			
	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Channel width (m)	3.1	± 1.8	1.7	5.1	2.3	± 1.1	1.1	5.5	3.2	± 0.9	2.2	3.8
Thalweg depth (m)	0.3	± 0.04	0.3	0.3	0.2	± 0.1	0.1	0.4	0.3	± 0.1	0.13	0.3
Shade (% canopy cover)	23.9	± 26.9	0.7	53.3	60.3	± 28.0	5.7	96.8	94.4	± 1.2	92.9	95.3
CPOM (m^2)	0	0	0	0.01	0.3	± 1.1	0	4.8	0.1	± 0.1	0.01	0.1
Wood (m^2)	0.01	0	0	0.01	0.1	± 0.4	0	2.0	0.01	± 0	0.01	0.02
Root complexes (m^2)	0.9	± 1.2	0	2.3	5.2	± 8.1	0	29.7	4.1	± 0.3	3.9	4.4
Length of undercuts (m)	27.6	± 15.1	15.5	44.6	18.4	± 9.00	1.7	38.8	22.2	± 20.7	2.5	43.8
Overhanging bank vegetation (%)	12.3	± 5.8	6.5	18.0	54.2	± 26.8	12.0	100	23.3	± 21.0	4.5	46.0
Bedrock substrate (%)	0	0	0	0	2.9	± 7.7	0	28.0	0	0	0	0
Boulder substrate (%)	6.7	± 7.0	0	14.0	12.9	± 11.9	0	38.0	14.0	± 6.9	10.0	22.0
Cobble substrate (%)	35.3	± 12.1	24.0	48.0	18.6	± 13.9	0	42.0	24.0	± 8.7	14.0	30.0
Gravel substrate (%)	41.3	± 18.1	28.0	62.0	31.5	± 16.9	2	70.0	40.7	± 4.6	38.0	46.0
Sand substrate (%)	10.7	± 7.6	2.0	16.0	14.0	± 8.4	0	32.0	17.3	± 7.0	10.0	24.0
Silt substrate (%)	6.0	± 8.7	0	16.0	16.8	± 18.9	0	70.0	4.0	± 3.5	0	6.0
Clay substrate (%)	0	0	0	0	3.3	± 8.2	0	34.0	0	0	0	0
Run habitat (%)	54.0	± 31.4	32.0	90.0	64.1	± 22.7	31.0	100	49.3	± 10.1	40.0	60.0
Riffle habitat (%)	36.7	± 32.1	0	60.0	26.3	± 19.1	0	60.0	38.7	± 8.3	32.0	48.0
Pool habitat (%)	9.3	± 1.2	8.0	10.0	8.0	± 8.6	0	29.0	10.0	± 8.7	4.0	20.0
Other habitat (%)	0	0	0	0	1.8	± 5.0	0	21.0	2.0	± 3.5	0	6.0
Macrophyte cover (%)	0.5	± 0.5	0	1.0	2.4	± 6.4	0	26.0	0	0	0	0
Sample reach gradient (m/m)	0.04	±0.03	0.01	0.1	0.1	±0.1	0.01	0.2	0.1	±0.1	0.02	0.2

3.3.3 Factors associated with instream habitat

PCA was carried out to determine which instream habitat variables explained dissimilarities in instream habitat among study sites. The first two axes explained 39.2% of the variation among sites while four axes accounted for a total of 60.9% variation (Table 3.10).

Table 3.10: Eigenvalues and percent variation for each principal component (PC) axis for instream habitat variables across 26 study sites.

PC axis	Eigenvalue	% Variation	Cumulative % Variation
1	4.3	20.6	20.6
2	3.9	18.6	39.2
3	2.3	11.1	50.3
4	2.2	10.6	60.9

The PCA plot of instream habitat with vector overlay (Figure 3.3) indicated the majority of the Pirongia and Whaingaroa sites associated with wider stream channels and more riffles, boulders and cobbles, consistent with steeper upstream gradients. The Whatawhata native forest site (WK1) was also located to the top left of the plot reflecting a wider channel and coarser substrates than the other sites at Whatawhata. The remaining sites at Whatawhata were more closely associated with finer substrates such as sand, silt, clay and gravel, as well as more run habitat and macrophyte cover. The Pirongia sites on the right hand side of the plot are KA1 and TP5 and the Whaingaroa sites are BC2 and OM1, all of which were characterised by these types of habitat.

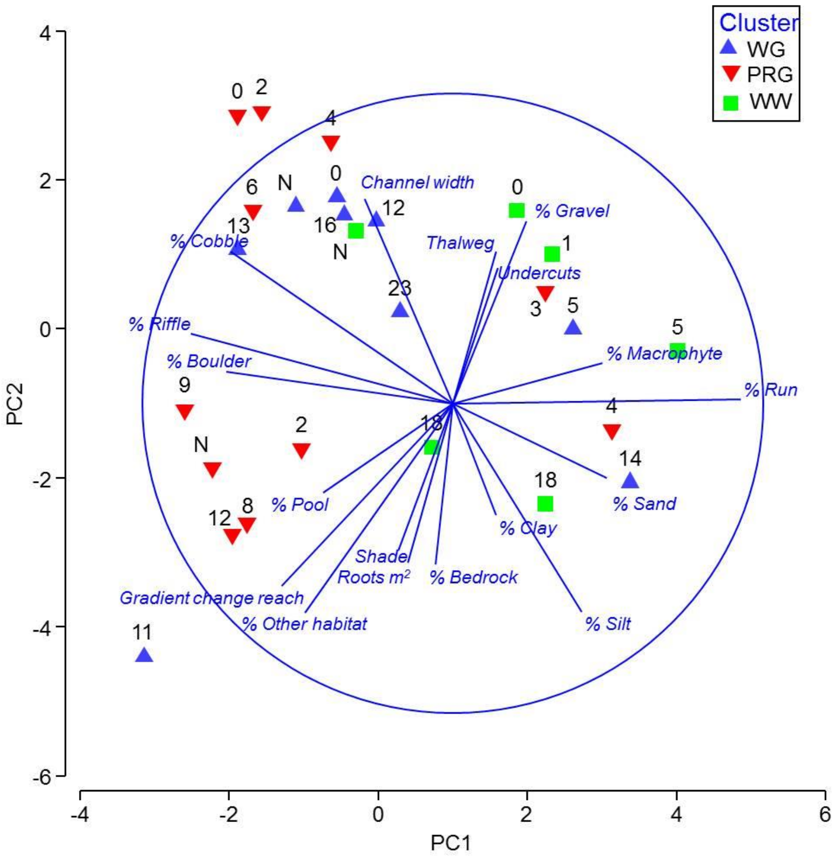


Figure 3.3: PCA plot based on instream habitat variables of all study sites by spatial cluster and riparian planting age along principal component (PC) axes 1 and 2, showing relationships with instream habitat variables with *r* >0.3. Clusters are: WG = Whaingaroa, PRG = Pirongia, WW = Whatawhata. Numbers refer to riparian planting age (years). Native study sites are marked N and pasture sites as “0”.

The first PCA axis represented predominantly hydraulic conditions and substrate size with percentage run/riffle habitat and percentage cobble or boulder substrate being strongly associated (Table 3.11). PC axis two associated with other hydraulic habitat types, percentage silt substrate and channel width. PC axis three mainly associated with riparian vegetation influences such as amount of wood (m^2), CPOM (m^2) and overhanging vegetation, while PC axis four was most strongly represented by bank undercuts, shade and macrophyte cover.

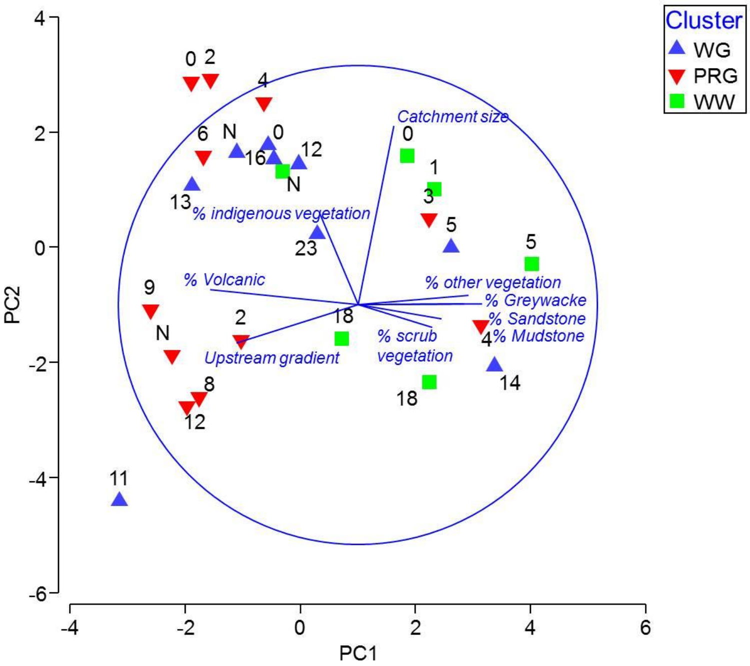
Table 3.11: Factor loadings of each habitat variables for the first four principal component (PC) axes for instream variables across 26 study sites. Values >0.3and <0.3 are shown in bold.

Variable	**PC1**	**PC2**	**PC3**	**PC4**
Channel width (m)	-0.137	**0.334**	-0.226	0.131
Thalweg depth (m)	0.067	0.247	-0.073	0.241
Shade (% canopy cover)	-0.084	-0.240	-0.234	**0.320**
CPOM (m^2)	-0.103	0.026	**0.533**	0.178
Wood (m^2)	-0.081	0.053	**0.536**	0.201
Root complexes (m^2)	-0.070	-0.260	-0.187	0.247
Length of undercuts (m)	0.070	0.221	0.115	**0.381**
Overhanging streambank vegetation (%)	0.035	-0.147	**0.367**	-0.160
Bedrock substrate (%)	-0.027	-0.262	-0.021	0.093
Boulder substrate (%)	**-0.351**	0.052	0.147	-0.154
Cobble substrate (%)	**-0.344**	0.247	0.015	-0.178
Gravel substrate (%)	0.115	0.297	-0.135	0.038
Sand substrate (%)	0.238	-0.122	-0.012	0.134
Silt substrate (%)	0.199	**-0.340**	0.194	0.129
Clay substrate (%)	0.067	-0.182	-0.115	-0.287
Run habitat (%)	**0.447**	0.006	0.007	-0.039
Riffle habitat (%)	**-0.407**	0.114	0.013	-0.038
Pool habitat (%)	-0.202	-0.146	0.024	**0.390**
Other habitat (%)	-0.230	**-0.341**	-0.041	0.008
Macrophyte cover (%)	0.233	0.065	0.193	**-0.343**
Reach gradient (m/m)	-0.266	-0.298	-0.058	-0.251

The PCA plot of instream habitat variables for sites showed little pattern with riparian planting age or upstream catchment variables represented in the overlay (Figure 3.4A). A group of sites that included various riparian planting ages, pasture and native sites were weakly associated with upstream catchment in indigenous vegetation. Another group of mainly Pirongia sites on the bottom left of the plot included various-aged-plantings and a native site, associated with more volcanic rock and higher upstream gradients. Whatawhata sites predominantly associated with “other” vegetation and geologies of greywacke, sandstone and mudstone. Pasture and several recently planted sites occurred across the top right of the plot and were characterised by larger catchment size.

The PCA plot for instream habitat variables with riparian overlay showed little pattern for site planting age or spatial cluster (Figure 3.4B). All vectors were short indicating the lower strengths of the relationships.

A



B

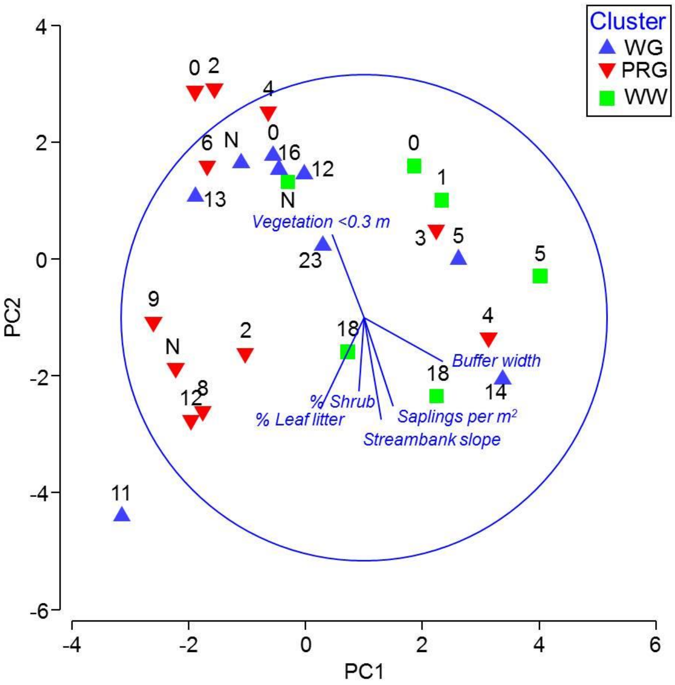


Figure 3.4A and B: PCA plots based on instream habitat variables of all study sites by spatial cluster and riparian planting age along principal component (PC) axes 1 and 2, showing relationships with upstream catchment (A) and riparian (B) variables with *r* >0.3. Clusters are: WG = Whaingaroa, PRG = Pirongia, WW = Whatawhata. Numbers refer to riparian planting age (years). Native study sites are marked N and pasture sites as “0”.

3.3.4 Variables explaining instream habitat dissimilarity

DistLM was used to examine which upstream catchment and local riparian factors best explained variation in overall instream habitat expressed as resemblance matrix between sites. Results from marginal tests are reported in Appendix 4. For upstream catchment variables, catchment (%) in greywacke was highly correlated with volcanic rock (r = -0.91) so was removed from the analysis. Upstream catchment in “other” geology consisted of small amounts of unconsolidated sands and gravels and alluvium materials with values <10%, so it was also removed from the analysis. No strong correlations (r >0.8) were present in the riparian data so all variables were included in the analysis.

Upstream catchment predictors

All three models using upstream catchment predictors contained the same variables except the forward selection process ranked the variables slightly differently to the manual model and accounted for 3% more variation than both the best-fit and manual processes. For all models, cumulative R^2 values were between 51 and 54% (see Table 3.12). For the final model using manual selection process (Table 3.13), a total of 51% (cumulative R^2) of the variation in instream habitat was accounted for by eight upstream catchment variables. Catchment geology dominated by volcanic rock and upstream catchment area accounted for almost half the variation in the model with cumulative R^2 values of 12% each. The remaining six variables accounted for between 3% and 5% variation each. Both percentage volcanic rock and upstream catchment area were statistically significant at p = 0.002 and 0.001, respectively. Although explaining 3% less variation than the forward selection model, the manual model was chosen as the most parsimonious model as it explained the same variation as the best-fit and contained upstream catchment variables in an order which provided a robust representation of the major catchment influences observed during GIS analysis.

Table 3.12: DistLM analysis summary of the cumulative effect of upstream catchment variables on instream habitat dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables which led to an increase in adjusted R^2 and for manual and forward selection processes contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Volcanic rock (%)	1	2	X
Catchment area (ha)	2	1	X
Indigenous vegetation (%)	3	6	X
Grass vegetation (%)	4	7	X
Scrub vegetation (%)	5	3	X
“Other” vegetation (%)	6	5	X
Distance to native forest (m)	7	4	X
Sandstone (%)	8	8	X
Total number of variables	*8*	*8*	*8*
Cumulative R^2	*0.51*	*0.54*	*0.51*

Table 3.13: Final model selected for the cumulative effect of upstream catchment variables on instream habitat dissimilarity among 26 study sites using manual selection. Only variables which led to an increase in adjusted R^2 and contributed >3% to the cumulative R^2 were included in the model. Significant p values (<0.05) are highlighted in bold.

Variable	**Adjusted R^2**	**SS(trace)**	**Pseudo-F**	**p value**	**Proportion. R^2**	**Cumulative R^2**
Volcanic rock (%)	0.08	61.36	3.18	**0.002**	0.12	0.12
Catchment area (ha)	0.17	62.02	3.55	**0.001**	0.12	0.24
Indigenous vegetation (%)	0.19	27.37	1.61	0.067	0.05	0.29
Grass vegetation (%)	0.21	25.75	1.55	0.108	0.05	0.34
Scrub vegetation (%)	0.22	21.69	1.33	0.236	0.04	0.38
“Other” vegetation (%)	0.24	24.07	1.51	0.127	0.05	0.42
Distance to native forest (m)	0.27	26.01	1.69	0.067	0.05	0.47
Sandstone (%)	0.28	18.19	1.20	0.279	0.03	0.51

Riparian predictors

The forward and best-fit models contained the same seven variables and both results accounted for 34% of the total variation. The manual model only contained six variables, accounted for less variation (29%), and differed from the other two results by containing percentage low vegetation ≤0.3 m. Results for all models are summarised in Table 3.14. Although accounting for 5% less variation than both the forward and best-fit models, the manual model was selected as the most parsimonious model to represent riparian predictor variables for instream habitat. It contained one less variable than the forward and best-fit models and provided a more intuitive representation of important variables associated with riparian dissimilarities between sites based on field work observations. The manual selection model is presented in Table 3.15.

A total of 29% (cumulative R^2) of the variation in habitat is accounted for by six riparian variables. Stream bank slope accounted for 6% variation, followed by average buffer distance (5%), number of saplings (5%) followed by basal area and leaf litter with 4%. Vegetation ≤0.3 m high accounted for an additional 5%. No p values were significant.

Table 3.14: DistLM analysis summary of the cumulative effect of riparian variables on instream habitat dissimilarities among 26 study sites using manual, forward and best-fit selection processes. Only variables which led to an increase in adjusted R^2 and for manual and forward selection processes contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Average stream bank slope (°)	1	4	X
Average buffer width (m)	2	6	X
Number of saplings (per m^2)	3	1	X
Basal area (m^2 per 20 m^2)	4	5	X
Leaf litter (%)	5	2	X
Vegetation ≤0.3 m in height (%)	6	-	-
Vegetation >0.3 m in height (%)	-	7	X
Shrubs (%)	-	3	X
Total number of variables	*6*	*7*	*7*
Cumulative R^2	*0.29*	*0.34*	*0.34*

Table 3.15: Final model selected for the cumulative effect of riparian variables on instream habitat dissimilarity among 26 study sites using manual selection. Only variables which led to an increase in adjusted R^2 and contributed >3% to the cumulative R^2 were included in the model.

Variable	**Adjusted R^2**	**SS(trace)**	**Pseudo-F**	**p value**	**Proportion. R^2**	**Cumulative R^2**
Average stream bank slope (°)	0.02	30.02	1.5	0.130	0.06	0.06
Average buffer width (m)	0.03	28.41	1.4	0.163	0.05	0.11
Saplings (per m^2)	0.05	26.13	1.3	0.208	0.05	0.16
Basal area (m^2 per 20 m^2)	0.05	21.54	1.1	0.348	0.04	0.20
Leaf litter (%)	0.06	23.62	1.2	0.277	0.04	0.25
Vegetation ≤0.3 m in height (%)	0.07	24.91	1.3	0.258	0.05	0.29

Combined upstream catchment and riparian predictors

Upstream catchment variables selected and total percent variation explained were similar for all upstream catchment models so all eight catchment variables presented in Table 3.12 were selected for further analysis. All eight riparian variables from the riparian models presented in Table 3.14 were also selected for further analysis. These riparian variables represented the results from all models and were selected as both the forward and best-fit solutions yielded the same results and all three models contained the first five variables.

Each resulting model contained ten variables which cumulatively accounted for between 59 and 61% of the total variation between sites. Seven of the ten variables were contained in all models and the best-fit model contained nine of the ten variables manually selected. Both the forward and best-fit solutions contained distance to native forest which differed from the manual results. Results for all models are summarised in Table 3.16.

For the final model using a manual selection process, a total of 60% of the variation in habitat dissimilarity among sites is accounted for by ten variables (Table 3.17). The dominant contributors to the model were upstream catchment area (12%) and upstream geology in volcanic rock (12%), both of which had significant p values. All other variables contributed between 3 and 6% of the variation within the model.

Table 3.16: DistLM analysis summary of the cumulative effect of selected upstream catchment and riparian variables on instream habitat dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables which led to an increase in adjusted R^2 and for manual and forward selection processes contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Catchment area (ha)	1	2	X
Volcanic rock (%)	2	1	X
Indigenous vegetation (%)	3	4	X
Number of saplings (per m^2)	4	3	X
Scrub vegetation (%)	5	10	X
Shrub (%)	6	-	X
Leaf litter (%)	7	-	-
“Other” vegetation (%)	8	-	X
Average stream bank slope (°)	9	7	X
Grass vegetation (%)	10	5	X
Vegetation >0.3 m in height (%)	-	6	-
Distance to native forest (m)	-	8	X
Buffer width (m)	-	9	-
Total number of variables	*10*	*10*	*10*
Cumulative R^2	*0.60*	*0.59*	*0.61*

Table 3.17: Final model selected for the cumulative effect of upstream catchment and riparian variables on instream habitat dissimilarity among 26 study sites using manual selection. Significant p values are in bold. Only variables which led to an increase in adjusted R^2 and contributed >3% to the cumulative R^2 were included in the model. Significant p values (<0.05) are highlighted in bold.

Variable	**AdjustedR^2**	**SS(trace)**	**Pseudo-F**	**P value**	**Proportion R^2.**	**Cumulative R^2.**
Catchment area (ha)	0.08	62.73	3.26	**0.001**	0.12	0.12
Volcanic rock (%)	0.17	60.65	3.47	**0.001**	0.12	0.24
Indigenous vegetation (%)	0.19	27.37	1.61	0.069	0.05	0.29
Number of saplings (per m^2)	0.23	34.56	2.14	0.014	0.06	0.35
Scrub vegetation (%)	0.24	18.80	1.17	0.278	0.04	0.39
Shrub (%)	0.24	18.93	1.19	0.302	0.03	0.42
Leaf litter (%)	0.27	25.46	1.66	0.086	0.05	0.47
“Other” vegetation (%)	0.29	24.20	1.63	0.096	0.05	0.52
Average stream bank slope (°)	0.31	20.55	1.42	0.139	0.04	0.56
Grass vegetation (%)	0.33	19.71	1.39	0.205	0.04	0.60

Final overall model selection for instream habitat

The most parsimonious and logical model selected to explain the variation in instream habitat for the combined upstream catchment and riparian variables was the best-fit solution summarised in Table 3.16, with a total of 61% of the variation explained between instream habitat among study sites. This model incorporated ten variables, seven of which were catchment derived and three of which were riparian derived. The best-fit solution differed from the manual solution by excluding leaf litter and including upstream distance to native forest which was considered mechanistically more informative. Although individual R^2 and adjusted R^2 values were not generated during best-fit analysis, the similarity of this model with the manual model was likely to reflect the strength and relative contribution of the individual components described therein.

3.3.5 Effects of riparian planting age on instream habitat

PERMANOVA revealed no significant effect of riparian age groups (Pseudo-$F_{4, 22}$ = 1.388, p = 0.119) or by spatial cluster (Pseudo-$F_{2, 22}$ = 1.313, p = 0.181) on instream habitat dissimilarity between sites. There was no significant interaction between these two factors (Pseudo-$F_{4, 22}$ = 0.732, p = 0.822).

Kruskal-Wallis testing of all individual instream habitat characteristics between riparian planting age groups revealed statistically significant differences in shade (H = 11.46, p = 0.02), CPOM (H = 10.66, p = 0.03) and percentage pool habitat (H = 10.53, p = 0.03).

Although pairwise comparisons for shade between age groups were not statistically significant, box plots indicate marked differences in means between pasture and recent riparian planting groups compared to mid, mid-late and old riparian groups (Figure 3.5).

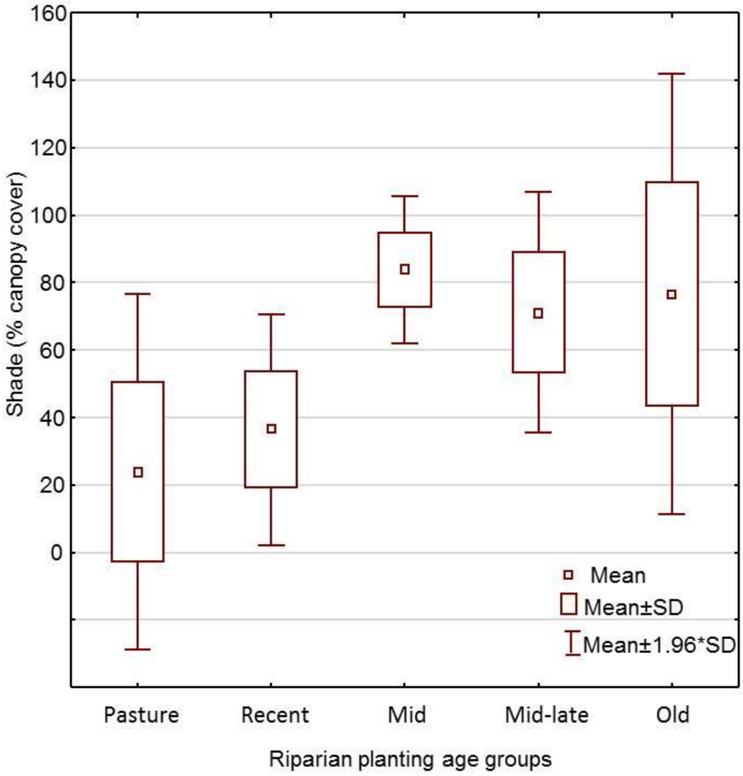


Figure 3.5: Box plot of shade (% canopy cover) by riparian age groups for 23 Pirongia, Whaingaroa and Whatawhata study sites.

Pairwise comparisons of CPOM also did not reveal any significant differences between riparian age groups (plot not shown). One site in the mid-late category (site AA1) had an unusually high amount of CPOM (4.8 m^2) compared to all other sites (<0.5 m^2) and is likely responsible for the significant result returned by the Kruskal-Wallis test.

For pool habitat, pairwise comparisons showed differences between recent and old sites (p = 0.04) indicating generally more pool area in the older riparian study sites (Figure 3.6).

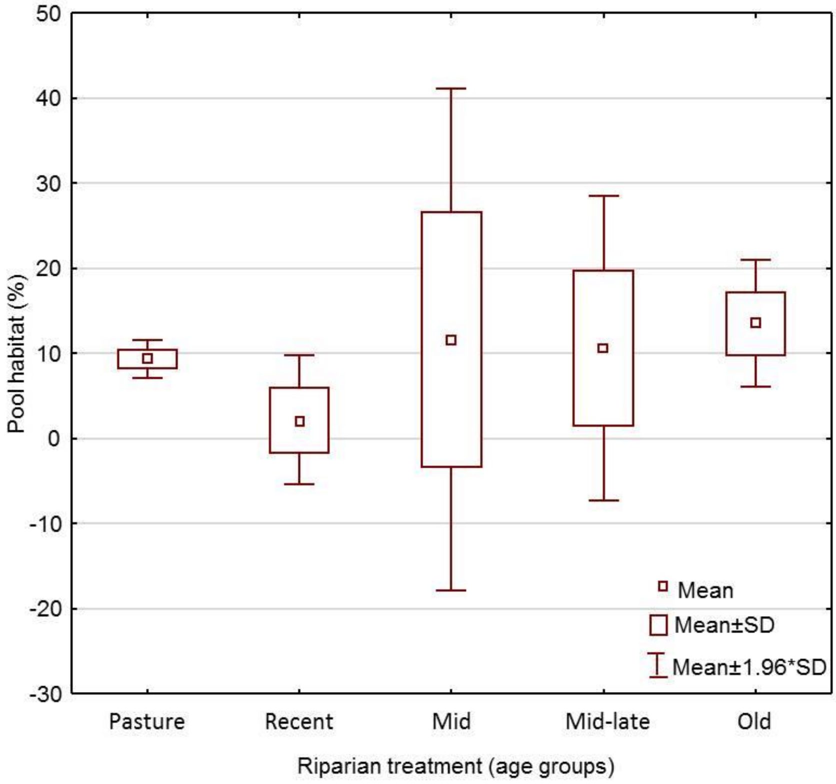


Figure 3.6: Box plot of pool habitat (% of reach) by riparian age groups for 23 Pirongia, Whaingaroa and Whatawhata study sites.

Mann-Whitney U tests for seven key instream variables between two riparian age groups and native sites revealed a significant shade difference between 11-15 years old plantings (mid-late) and native forest sites which typically had higher shade (p = 0.03). However, tests revealed no statistically significant difference between old riparian sites (16+ years) and native sites for any of the variables tested.

To further explore changes in shade over time since planting, mean shade was plotted by riparian site age (in Figure 3.7). Shade levels of 70% are indicated by the orange line, which is the minimum threshold for achieving water temperature reduction and is above the 60% target for exerting influence on algal blooms and changes to invertebrate communities (Rutherford *et al.*, 1999). The plot trend line (R^2 = 0.39) indicates a general trend for shade to increase with riparian age, with 70% canopy cover seen in some sites as early as six years on from planting and fencing. The lower shade values for one of the 12 year old riparian planting site (TA1) were associated with large areas of rank grass within the riparian area, while the 13 year old site (AA1) and the 16 year old site (TM1) (see Figure 2.1 for locations) were associated with a high proportion of flax in the riparian margins compared to other sites.

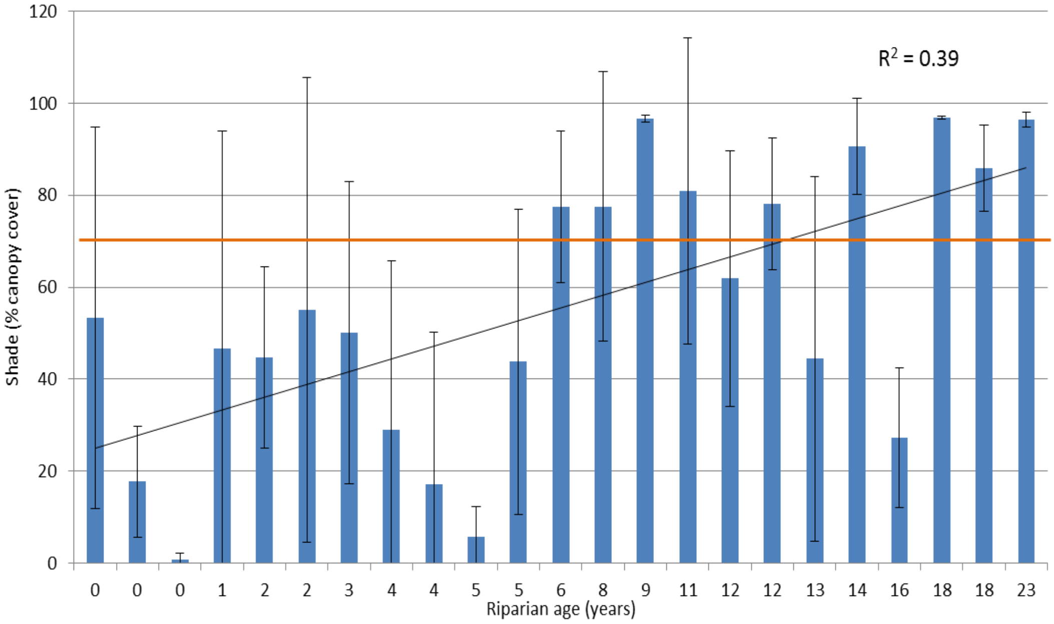


Figure 3.7: Mean shade (% canopy cover) by riparian age (years) ± SD for 20 riparian sites and 3 pasture sites in Pirongia, Whaingaroa and Whatawhata. 70% shade cover is defined by orange line with linear trend line in black and associated R^2 value.

Channel width was also plotted against riparian age to test the hypothesis that pasture channels widen as they are retired as they move on a trajectory towards native forest characteristics (Davies-Colley, 1997). Results were inconclusive due to a wide range of different sized streams (1.1 to 5.4 m) and the inability to compare reaches inside and outside the riparian planting which were not part of this study design.

3.4 Discussion

The focus of this chapter has been on: i) characterising instream habitat among small stream sites in the western Waikato that differed primarily in terms of riparian vegetation planting age; and ii) determining the relative importance of upstream catchment and/or reach-scale riparian factors on instream habitat dissimilarity among sites. All riparian planting sites were fenced, and along with the native forest reference sites were not subject to stock grazing. Pasture sites were unfenced and accessible to stock, and were included to represent conditions that prevailed prior to riparian fencing and planting. Sites were clustered across three areas each with native forest and pasture sites represented. The habitat features measured were considered relevant to kōura populations. Associations between kōura and instream habitat are explored in Chapter 4.

3.4.1 Reach-scale instream habitat characteristics

The streams sampled in this study were generally typical of wadeable Waikato hill-country streams such as those reported in Collier *et al.* (2007). Average stream channel widths were between 1.1 and 5.6 m with average thalweg depths between 0.1 and 0.4 m. Sites were characterised by a mix of substrate sizes ranging from predominantly boulder and cobble to sites where gravel, sand and silt were prevalent. Hydraulic conditions ranged from relatively steep, riffle, dominated streams, to lower gradient sites with more run habitat enabling macrophytes to establish.

With the exception of one site, all streams had confirmed presence of kōura. Kōura have previously been found in streams with different catchment land-use such as those located in native or exotic forest as well as pasture (Hicks & McCaughan, 1997; Parkyn *et al.*, 2002), all of which are represented in either study reaches or upstream land-use at the study sites. A key feature influencing kōura abundance across land-use is instream cover as previously reported in Parkyn and Collier (2004), which includes habitat such as tree roots, leaf litter, undercut banks, woody debris and macrophytes, all of which varied in both presence and quantity between study sites. Jowett *et al.* (2008) found the majority of kōura in North Island streams in depths between 0.2 and 0.3 m and observed the probability of kōura presence to be reduced in streams over 6 m wide.

Streams in the present study were all below 6 m wide, with the majority less than 3 m in width and most had depths suitable for optimal kōura capture.

In this study, a number of key factors were found to structure instream habitat dissimilarities between study sites. These included channel factors such as width, hydraulic conditions, substrate type and bank undercuts, as well as variables associated with the influence of riparian vegetation such as cover by CPOM, wood and root complexes. These factors were all considered important for predicting kōura occurrence and habitat preferences by Jowett *et al.* (2008), confirming their general significance across a range of stream types supporting kōura.

There were some notable differences in instream habitat between different clusters of sites. Pirongia sites were relatively homogenous and were characterised by riffle and pool habitat with a dominance of boulder and cobble substrate. The exceptions were two sites (KA1 and O1) which had a greater similarity to Whatawhata sites which were characterised by a dominance of run habitat and smaller substrates such as gravel, sand, silt and clay, interspersed with some bedrock and cobble areas.

Whaingaroa sites had the highest degree of heterogeneity amongst the clusters. Six of the Whaingaroa sites were relatively similar and dominated by cobbles and wider channels, however, the steep gradient and cascade habitat at BC1 along with differing substrate and habitat types at OM1 and BC2 were drivers for the heterogeneity within this site cluster.

Whatawhata sites were generally characterised by smaller substrates such as gravel, sand, silt and clay, interspersed with some bedrock and cobble areas. Most had more run habitat and macrophytes than those in the Pirongia or Whaingaroa clusters.

A number of key upstream catchment mechanisms were associated with the spatial distribution of sites in relation to instream habitat in this study. Upstream catchment size, geology and, to a lesser degree, upstream catchment land-cover were the strongest influences on habitat. Richards *et al.* (1996) found a similar pattern in streams sampled in Michigan, USA with catchment size and geology primary drivers of differences in instream habitat followed closely by differing upstream land-use. The spatial distribution of sites in this study is driven by clear differences in geology found at Whatawhata compared to both Whaingaroa and Pirongia, coupled with a wide range of upstream catchment sizes between sites.

Catchment land-use has previously been identified as an important influence in shaping instream habitat, particularly the extent of agricultural land present in the upstream catchment and its influence on hydrology, channel characteristics as well as sediment delivery (Allan *et al.*, 1997) and water quality (Davies-Colley, 2013). Land-use influences can be scale-dependent and their impact on instream habitat at a reach-scale can often be difficult to quantify (Lammert & Allan, 1999). Spatial relationships between instream habitat and upstream land-cover for this study were relatively weak but did illustrate key differences between various catchments and dominant land-cover categories. The prevalence of indigenous vegetation upstream of many sites on the flanks of Mount Pirongia and Karioi and the presence of exotic forestry and scrub in many of the Whatawhata sites were evident in the PCA plots of instream habitat.

PCA plots of riparian variables did not have any apparent pattern in the spatial arrangement of sites, either by riparian age or site spatial cluster, for instream habitat dissimilarities and the riparian overlay showed weak relationships. Patchiness of both basal area and number of saplings varied between sites and did not reflect the expected upward trajectory (and therefore reduced patchiness) of both variables as riparian planting aged. Major differences in riparian management techniques and their implementation between sites were observed during field work. These differences included buffer width, plant species and planting density, as well as extent of remnant native vegetation present at sites, and may, at least in part, be responsible for the observed lack of any strong relationships.

Despite apparent spatial differences in geology, land-cover and some habitat characteristics, PERMANOVA results indicate that overall instream habitat dissimilarity between site spatial clusters was not statistically significant. This finding provides evidence that, collectively, the streams selected for this study provide a level of instream habitat homogeneity suitable for discerning riparian vegetation influences on kōura populations (see Chapter 4). As such, the results would be broadly applicable to other wadeable, hill-country North Island streams.

3.4.2 Upstream catchment and riparian factors influencing instream habitat

Upstream catchment predictors accounted for just over half the variation in instream habitat between study sites when analysed using DistLM. The two most important factors were the size of the upstream catchment and the presence of volcanic rock geology, followed by various classes of upstream land-cover and site distance to native forest. Strong similarities with these results are seen in Michigan (USA) streams where Richards *et al.* (1996) also found approximately 50% of variation in physical habitat was accounted for by landscape-scale variables, with the majority being a function of geology-structure variables, such as catchment size and rock type.

Upstream catchment size exerts influence on the dimensions of the stream channel (Richards *et al.*, 1996) as well as regulating hydrological effects on instream habitat through flow volume, flow variability and water velocity (Jowett *et al.*, 1996; Allan & Castillo, 2007), all of which have been identified as key predictors of kōura presence and abundance (Parkyn & Collier, 2004; Jowett *et al.*, 2008). Upstream catchment sizes ranged from 18 to 794 ha which, coupled with upstream gradient differences (0.006 m/m to 0.15 m/m), appeared to exert strong influences on instream characteristics through flow regimes, although flow data were not available for this study.

Upstream catchment geology influences river morphology, substrate type and water chemistry, and can moderate influences of hydrological variation (Richards *et al.*, 1996; Weigel *et al.*, 2003; Lintern *et al.*, 2018). Distinct geology types exist between many of the study sites, and can be expected to have influenced stream hydraulic conditions and substrate type. The sedimentary rocks found around Whatawhata such as greywacke, sandstone and mudstone are prone to erosion and weak in structure (Selby & Lowe, 1992) resulting in smaller stream substrates. Volcanic rocks such as compacted lava and pre-Taupō eruption ashes derived from the basalt volcanoes of Karioi and Pirongia consist of welded rock resulting in coarser stream substrate such as cobbles and boulders, with some sand and silt inputs from more recent ash deposits (Waterhouse & White, 1994). Given the geographical proximity of all the study sites, these mixed dominant bedrocks likely led to the importance of upstream geology in explaining variation in habitat dissimilarity between sites.

Previous studies have attempted to establish links between upstream land-use and stream habitat, however, this is often confounded by historic land-use, topography, differing soil types, varying extents of riparian vegetation, as well as other anthropogenic modification such as channelisation and hydrological alteration (Richards *et al.*, 1996). Allan *et al.* (1997) found the extent of agricultural land at a sub-catchment level was the best single predictor for local stream conditions in Michigan, USA, which was supported by Richards and Host (1994) in regard to nutrient and sediment inputs to streams increasing as agricultural land-use increased in the upper catchment. Although changes in water quality have not been the focus of this study, differences in upstream land-cover accounted for close to 20% of the variation in instream habitat between study sites, and was driven by the amount of indigenous forest, grass, scrub and other land-cover present upstream of the sample sites. The extent of indigenous forest, scrub and exotic forest can influence streams by mitigating overland flow and reducing flow variability, as well as providing upstream sources of detritus and wood inputs to streams, and reducing fine sediment and water temperatures, particularly when associated directly with riparian zones (Hanchet, 1990; Quinn *et al.*, 2009). Indeed, intact indigenous forest in the headwaters of many of the sites, coupled with distance to contiguous native forest, were also likely to be important in determining habitat stability which is critical for providing stable refugia for kōura during high flow events (Parkyn & Collier, 2004).

Interestingly, reach scale riparian factors had little bearing on instream habitat dissimilarity between sites, and at best accounted for 29% of the variation when analysed using DistLM. No one riparian factor dominated and, with the exception of stream bank slope, all were associated with vegetation characteristics expected under different-aged riparian plantings, such as number of saplings, basal area, leaf litter, shrub cover and low (≤0.3 m) vegetation. Stream bank slope is influenced by channel morphology as well as by the topography and gradient of the stream valley (Mosley, 2004), and its relevance in the variation explained in this study is more to do with other upstream catchment variables such as geology, upstream catchment size, topography and stream gradient rather than directly influenced by riparian condition. Nevertheless, the influence of bank slope has previously been found to dictate shade in small streams (Quinn *et al.*, 1997), and may well influence the provision of organic matter inputs to streams.

Where the effect of riparian plantings on stream communities has been evaluated, their success has been mixed and highly dependent on stream-specific factors, such as upstream catchment influences including geomorphology, along with buffer age and width (Parkyn *et al.*, 2003; Burrell *et al.*, 2014) . Holmes *et al.* (2016) found riparian fencing resulted in sediment reduction to a South Canterbury stream but that riparian vegetation had minimal direct influence on instream habitat in the sampled areas. Indeed, both Allan *et al.* (1997) and Lammert and Allan (1999) found catchment land-use more strongly influenced biotic integrity and habitat conditions at a reach scale than riparian condition, which seems to be the case in the present study.

When upstream catchment and riparian variation were examined together, upstream catchment factors still explained 42% of the variation compared with 20% explained by riparian variables, illustrating the strength of influence by catchment-scale factors. Results from these analyses support Hypothesis 1 (see Section 3.1) that catchment characteristics would significantly influence reach-scale habitat, with catchment size, land-cover and geology exerting the strongest influence. Additionally, findings in this study, support the notion presented by Frissell *et al.* (1986) and Winterbourn (1981) of a hierarchy of nested spatial levels with streams, each constraining the level below. At the largest scale, catchment-scale variables appear to be exerting more influence on instream habitat than local riparian factors in the present study.

3.4.3 Effect of riparian planting age on instream habitat

Increased shade with planting age was confirmed in this analysis, with marked changes in canopy cover between plantings ≤5 years old compared to older plantings (≥11 years). For the sites in this study, at least 70% shade (% canopy cover), the minimum threshold for achieving water reduction (Rutherford *et al.*, 1999) occurred with plantings at least 12 years old, however, some smaller sites achieved this sooner. Only spot temperature measurements were collected for this study so the effect of shading on stream temperature, over time, at these sites is unknown.

The level of shade did depend on the type of vegetation present at the study sites. A prevalence of flax at some sites seemed to exert less shade influence than shrubs or woody vegetation. Variation in shade between sites was a function not just of riparian planting age, but also stream bank slope and the presence of grass and/or occasional shrubs close to the stream edge. Quinn *et al.* (1997) also found these features reduced incoming diffuse solar radiation by an average of 71% in pasture sites in Waikato hill-country streams. Additionally, at some sites, the presence of one or two large trees created extensive shading.

The lack of any statistically significant differences between all riparian planting age groups for all key variables (with the exception of shade, pool habitat and CPOM) indicates that, at these study sites, riparian planting age had minimal effect on kōura habitat, and as such, findings do not support Hypothesis 2 (see Section 3.1). Although this was surprising, the ability to quantify the effect of riparian planting and fencing on instream habitat and invertebrate metrics can be problematic, as noted in previous studies such as Parkyn *et al.* (2003). In the present study, the influence of upstream catchment characteristics appeared to exert over-riding influence on instream habitat providing a reasonable explanation as to why differences between different riparian age, pasture and native sites were not more pronounced. The lack of differences may also relate to the proximity of many riparian and pasture sites to the moderating influence of upstream native forest, and their location in the upper reaches of catchments which is often characteristic of better quality habitat. These attributes may have reduced our ability to differentiate the influence of riparian planting age on instream attributes due to the existing relatively “good” habitat in this study.

With the exception of shade, the similarity of sites with riparian plantings ≥11 years old and native forest sites for key kōura habitat variables supports the findings from other studies that influences from riparian planting, at least for changes in invertebrate communities, can be realised in relatively short periods of time (Quinn *et al.*, 1997; Jowett *et al.*, 2009). For shade, results for comparisons between 11-15 year old plantings (mid-late) and native sites have been complicated by lower values at two sites (AA1 and TA1; see Figure 2.1). AA1 was dominated by flax plantings which appear to provide poor shade and TA1 was characterised by some large, open areas in rank grass. As previously discussed, sites in general followed a trajectory of increased shade with riparian planting age, and sites with 16+ year old plantings showed no statistically significant difference when compared to shade at native forest sites, despite very low shade at TM1 (16 years old) which again, was almost entirely dominated in flax plantings. Providing additional support to the above findings, PERMANOVA results also indicated that overall, instream habitat dissimilarity between sites of different riparian planting age was not statistically significant

Although I did not find clear patterns in relation to riparian planting age group because of overriding catchment and landscape influences, older riparian sites did appear to provide instream habitat similar to native sites, supporting Hypothesis 3 (see Section 3.1) that older riparian plantings 16+ years old will closely resemble native forest sites in terms of shade and important kōura habitat variables such as root complexes, woody debris, CPOM, undercut banks and overhanging vegetation. In fact, these similarities may occur as early as 11 years after planting in small streams.

Previous observations by Davies-Colley (1997) and predictions in Collier *et al.* (2001) indicate incised pasture streams are likely to widen as shade increases through riparian planting, and reconfigure to a morphology similar to native streams. The presence of more pools in old riparian sites compared to recent sites could be a function of channel widening and changes in channel morphology. The channel widening theory was investigated in relation to riparian planting age in the present study, however, results were inconclusive because the study was not designed specifically to test this hypothesis. The removal of streams over 3 m did not add any clarity to the analysis, and it was not possible to compare riparian reaches to both pasture and native within the same sub-catchment.

Chapter 4

Influence of upstream catchment, riparian and instream habitat characteristics on *Paranephrops planifrons* distribution, density and size

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4.1 Introduction

Paranephrops planifrons plays a significant role in New Zealand stream ecosystems by influencing: i) energy transfer between terrestrial and aquatic systems, and within lentic and lotic ecosystems (e.g., Jones et al, 2016); ii) invertebrate community structure and biomass (Parkyn *et al.*, 1997); iii) macrophyte density (Jurcak *et al.*, 2016); and iv) prey availability for both aquatic and terrestrial predators (Englund & Krupa, 2000). They have been referred to as a keystone species (Collier *et al.*, 1997), a term used to characterise organisms that exert a disproportionately large effect on an ecosystem relative to their abundance or biomass through their ability to influence community structure, mediate the transfer of trophic resources and modify the environment (Payton *et al.*, 2002). Kōura are also considered a cultural keystone species, prized by Māori and integral to maintaining connections with traditional practices (Noble *et al.*, 2016).

Preferred habitats reported for *P. planifrons* are tree roots, leaf litter, undercut banks and fallen logs/woody debris, particularly where this material from riparian sources is associated with banks alongside pools or forms debris clusters (Parkyn & Collier, 2004; Jowett *et al.*, 2008; Clearwater *et al.*, 2014). In pasture streams, where riparian tree cover is often lacking, kōura commonly use cobbles and instream vegetation such as grass and macrophytes (Parkyn, 2000). For young-of-the-year (YOY) kōura, stream edge habitat, low velocity shallow areas and fine sediment are known to provide important refugia and cover, while adults are often associated with cobble substrate (Olsson *et al.*, 2006; Jowett *et al.*, 2008). Indeed, both Burskey and Simon (2010) and Chucholl and Schrimpf (2016) found cover availability as the most consistent predictor of crayfish abundance at the reach-scale in USA and Germany, respectively.

Catchment-scale influences have also been identified as important determinants of crayfish presence. Water quality and chemistry are influenced by catchment geology and hydrology, historic and contemporary catchment land-use and vegetation cover (Lintern *et al.*, 2018). Water conductivity, pH, dissolved oxygen and calcium content have been reported to influence crayfish persistence in streams (Smith *et al.*, 1996), where higher calcium concentrations can decrease the time taken for exoskeleton hardening, thereby reducing predation vulnerability (Hammond *et al.*, 2006). Olsson *et al.* (2006) reported that *P. planifrons* was associated with streams where calcium concentrations were as low as 0.9 mg/L, indicating that high dissolved calcium is not a pre-requisite for crayfish presence. Nevertheless, both Burskey and Simon (2010) and Magoulick *et al.* (2017) found conductivity to be a positive predictor of crayfish presence in two separate studies in the USA, and this may partly reflect calcium concentrations.

Catchment area influences stream size and instream habitat characteristics such as depth, velocity and substrate (Jowett *et al.*, 1996), and interacts with topography, geology and hydrology to shape other stream habitat attributes (Allan & Castillo, 2007). For example, Burskey and Simon (2010) found the increased quality of run and riffle sequences along an Indiana, USA, stream was a negative predictor of crayfish presence. In a study of 30 wadeable, North Island rivers, Jowett *et al.* (2008) found the probability of finding *P. planifrons* declined once streams became wider than 6 m and substrate became dominated by boulders (>256 mm across); they recorded the highest number of crayfish in still or slow flowing water with velocities <0.4 m/s. Magoulick *et al.* (2017) found the landscape variables of stream order, geology, slope, topography and land-use as important determinants of crayfish occupancy in streams in Missouri and Arkansas, USA. Similarly, Mouser *et al.* (2019) found presence of some *Faxonius* spp. in the Ozark Highland ecoregion in the USA dependent on cumulative conditions in the upstream catchment area, notably catchment-scale geology along with stream elevation which collectively influenced nutrient concentrations, hydrology, substrate particle size, water temperature and suspended sediment levels. In west-central Indiana, USA, Burskey and Simon (2010) also found crayfish abundance was strongly predicted by catchment-scale variables with the presence of riparian forest a positive predictor and catchment size a negative predictor.

Riparian fencing and planting are used as tools to mitigate pastoral land-use effects on streams and have been noted to provide benefits to instream habitat, particularly key variables associated with kōura occurrence. Increased shade provided by riparian cover moderates water temperature and regulates primary productivity by algae (Davies-Colley & Quinn, 1998) and macrophytes (Rutherford *et al.*, 1999; Quinn *et al.*, 2007), while assisting in reducing diurnal variation of pH and dissolved oxygen in streams (Wilcock *et al.*, 1999). *Paranephrops planifrons* has been associated with streams with pH ranging between 4.1 and 7.9 (Olsson *et al.*, 2006) but water temperature is the primary determinant of growth and influences crayfish movement and dispersal (Parkyn *et al.*, 2002; Jones *et al.*, 2016). Parkyn *et al.* (2002) examined differences in kōura population structure in Waikato hill-country streams under different land-uses and found similar annual production between native forest and pasture sites, but greater densities in native forested sites. This finding reflected faster growth of kōura in pasture streams, primarily driven by differences in temperature, so that they reached maturity earlier but had reduced longevity compared to native forest populations.

Major factors differing between pasture sites and those with intact native vegetation include presence and amount of woody debris, substrate coarseness and water temperature (Hanchet, 1990). Riparian vegetation can influence instream habitat through provision of woody debris, coarse particulate organic matter (CPOM), root complexes, undercut banks and cover from overhanging stream vegetation (Parkyn *et al.*, 2000; Quinn *et al.*, 2009), which can provide refugia for crayfish from flow extremes and predation (Johnson *et al.*, 2003). In Waikato hill-country streams, Parkyn and Collier (2004) found native forest provided more stable instream habitat than a pasture site during an extremely high flow event, with kōura numbers remaining similar before and after the flood event at the native forest site. The dominance of cobbles and macrophyte habitat in pasture streams did not provide stable refugia during this event, while undercut banks, tree roots and pools associated with the riparian zone in native forest streams were associated with habitat stability. March and Robson (2006) also found that patches of forest or vegetated areas within agricultural catchments were associated with stable stream habitat for crayfish during high flow events, thereby providing a refugia for repopulation to downstream areas. Provision of stable refugia may be an important variable for *P. planifrons* population persistence given their weak dispersal abilities, as indicated by the high degree of inter-catchment genetic dissimilarity (Smith & Smith, 2009).

Responses of streams to riparian fencing and planting occur over a variety of timeframes. In pasture streams at Whatawhata, reductions in water temperature were observed as early as six years after fencing and native planting in small streams 1-2 m wide (Quinn *et al.*, 2009), while pine afforestation created shade close to levels found in native forest streams within 15 years of planting (Quinn *et al.*, 1997). At the same site, macroinvertebrate community indices increased within eight years in response to restoration of riparian vegetation (Jowett *et al.*, 2009).

Although responses to riparian plantings are well documented for macroinvertebrate assemblages, specific changes to kōura abundance and distribution have not been examined. Kōura are often missed during surveys targeting fish or invertebrates as most conventional sampling techniques are not effective at crayfish capture (Collier *et al.*, 1997), and provide unreliable estimates of crayfish density (Giling *et al.*, 2009). Several studies have reviewed various crayfish sampling methods and their effectiveness, showing that the efficiency of capture is a function of behaviour, habitat preferences and instream characteristics (Price & Welch, 2009; Parkyn, 2016). For *P. planifrons*, electric-fishing was found to be the most effective sampling method for use in streams with undercut banks and heavy cover (Rabeni *et al.*, 1997). Similarly Price and Welch (2009) found electric-fishing the most effective tool for documenting species richness and for collecting the greatest number and size range of diverse crayfish communities in North American streams, although in larger substrate types crayfish were able to avoid capture by hiding under rocks.

The importance of water flow was highlighted by Larson and Olden (2016) using various sampling techniques to increase capture of crayfish into the net through movement of water. Alonso (2001) found the catchability of crayfish increased with body size using electric-fishing, but small crayfish (<40 mm total length) still comprised close to one third of the total sample catch. *Paranephrops planifrons* is known to exhibit tail flipping behaviour when first in contact with electric current before quickly seeking cover under rocks, undercuts or other habitat which can make them difficult to capture consistently (Hicks, 2018).

To reduce electric-fishing inefficiency for collecting small kōura and potential escape of larger kōura among coarse substrates, the present study used a combination of electric-fishing with substrate movement and sweeping. This approach combines electric-fishing with aspects of kick-net sampling to disturb habitat, as well as providing a mechanism to create flow and increase the potential to capture kōura in the downstream stop-net.

Using this sampling approach, the aim of this chapter was to determine the relative influence of upstream catchment, riparian and instream habitat factors on kōura populations in Waikato hill-country streams with contrasting riparian vegetation to understand the role of riparian planting in mitigating land-use impacts on kōura.

Specific objectives for this chapter are:

- Determine the relationship between measured water quality and habitat factors and kōura density and size;
- Establish which catchment and riparian factors best explain variation in kōura density and size amongst sites;
- Determine the effects of riparian planting age on kōura density and size compared to streams in pastoral (no riparian planting) and native forest land-cover.

Based on published work, the following hypotheses were tested in this chapter:

1) Instream cover (e.g., root complexes, wood, undercut banks, CPOM) will positively influence kōura density, while hydraulic stress (e.g., frequency of riffle and cascade habitat with larger substrates) will have a negative impact on kōura density.

2) Differences in kōura density among sites will be influenced more by catchment-scale variables that control substrate size and hydraulic habitat (e.g., gradient, geology) than local-scale riparian planting.
3) Abundance of kōura, including young-of-the-year (YOY), will be lower in pastoral or early riparian sites than in older riparian planting sites which will be similar to native forest sites.

4.2 Methods

Sampling was undertaken within Department of Conservation Electric-Fishing Permit – 67878-RES and authorisation from Auckland/Waikato Fish and Game Council. For sites within Crown land administered by the Department of Conservation, sampling was authorised through Research and Collection Permit 70894-FAU.

Instream habitat measurements, such as CPOM, wood and macrophyte cover which may have been compromised by electric-fishing disturbance were collected prior to kōura sampling. Riparian and the remaining instream habitat measurements were collected after kōura sampling had concluded, as described in Chapter 3.

4.2.1 Measurement of physicochemical variables

Water temperature (°C), dissolved oxygen (mg/L and % saturation), pH, conductivity (µS/cm adjusted to 25°C) and turbidity (NTU) were measured at the downstream end of each sample reach using an YSI EXO ProDSS Handheld device fitted with an EXO1 sonde (Yellow Springs Instrument, Yellow Springs, USA). A 250-mL water sample was also collected in a polyethelene bottle at each site and analysed for water hardness at NIWA Hamilton using the EDTA titrimetric method based from Method No. 2340c in American Public Health Association (1989). Hardness is expressed as mg/L of calcium carbonate ($CaCO_3$). With the exception of site TU2 (see Figure 2.1), all measurements and sample collection were conducted between 8 am and 10.30 am on the day of sampling. TU2 measurements were collected at 12.30 pm. Physicochemical measurements were collected at arrival on site, prior to any stream disturbance.

4.2.2 Kōura sampling and measurement

Depletion methods for estimating population numbers are based on achieving >75% reduction (from the previous pass) which typically involves between two and four consecutive passes (Armour *et al.*, 1983). Rabeni *et al.* (1997) found very little difference in abundance estimates for *P. planifrons* between five fishing passes compared to two. For this study, an initial trial at a representative site indicated the majority of kōura (>75%) were captured in two compared to three electric-fishing passes (Kelly, 2018). Given the time required to effectively fish each 50-m sample reach, and since the aim was to provide a standard estimate of density per unit effort rather than a population estimate, two consecutive passes were considered adequate to provide data on kōura abundances for comparison among sites.

Each 50-m sample reach was divided into 5 x 10-m sub-reaches which were electric fished in two consecutive passes using a Kainga 300 Electric-Fishing Machine (EFM) combined with substrate disturbance and brushing using a wooden-handled, soft bristled broom. The exception was BC1 (see Figure 2.1) where the reach was 43.5 m long due to the presence of large boulder areas which were not able to be fished, resulting in sub-reaches between 5 and 10 m in length. Prior to fishing, stream conductivity was measured and best practice for voltage settings was used to optimise fishing effort while considering fish welfare (Joy *et al.*, 2013). Accordingly, pulse frequency for all sites was 60 pulse/sec with a pulse width of 2 m/sec which is commonly used to target small fish and was therefore considered the most likely to be effective for capturing crayfish. Anode size (small or large) varied between sites and was a function of stream type and habitat.

Fishing was completed at each site within a day with the exception of O1 which was fished over two consecutive days. Sampling was primarily conducted by a team of four comprising one fisher, one person on a stop-net and two team members sweeping to dislodge kōura and create flow into the downstream hand-held stop net. In smaller streams, or where habitat was less complex and space was constrained, fishing was occasionally conducted using three people with only one team member sweeping while the fisher also utilised a broom to create flow behind the EFM anode. A net (3 mm x 3 mm mesh) was installed across the downstream end of each sub-reach prior to being sampled to capture any kōura that avoided hand nets.

Areas of 1-1.5 m^2 were fished in bands moving progressively upstream across the width of the channel. Sweeping occurred during shocking and also for approximately 20 sec after shocking had ceased. In areas with predominantly cobble or gravel substrate, sweeping was supplemented with substrate disturbance by feet or hands after shocking was complete. The handheld stop-net was checked after each area within a sub-reach had been fished and all kōura were placed in buckets on the stream edge. The downstream fixed net was checked after two consecutive passes had been completed and any kōura were combined into the relevant sub-reach sample. No bias was placed on any particular habitat during sampling with equal effort applied across all sample reaches.

Orbital carapace length (OCL mm) of each kōura was measured with manual callipers from the back of the eye socket to the mid-dorsal posterior border of the carapace (Hopkins, 1967a). All kōura were sexed unless too small to identify gonads in which case they were recorded as “unidentified sex”. Following Parkyn (2000) and Hopkins (1967b), female kōura were considered mature at ≥17 mm OCL, and kōura ≤13mm were considered YOY which aligns with Parkyn’s (2000) assessment of growth rates in Waikato hill-country streams in both native and pasture land-cover. Any kōura in berry or with attached young were also recorded. Following measurement, all kōura were returned to the source sub-reach after the downstream net had been moved to the next upstream sub-reach to prevent movement and double counting.

4.2.3 Statistical analyses

At the start of the analysis, the relationship between kōura (no. per m^2) versus kōura (no. per 100-m bank length) was tested using Spearman rank non-parametric tests. These two variables had a strong correlation (r_s = 0.92) so kōura (no per m^2) sampled was selected for subsequent analysis to be consistent with data presented in relation to fish in Chapter 5. The relationship between mean kōura length and median kōura length was also and found to be strongly correlated (r_s = -0.98) so mean kōura length per reach was used for further analysis. Spearman rank correlations were also used to examine the strength and direction of association between key instream habitat variables and total kōura density (no. per m^2).

Factors affecting kōura abundance

Step-wise distance-based linear models (DistLM, PRIMER 7.0.13) were used to test which combination of upstream catchment, riparian and instream habitat variables (see Chapter 3), and physicochemical variables, best explained the dissimilarities in kōura density (no. per m^2) between sites and created the most parsimonious model. All sites were entered into the analysis (i.e., native, riparian and pasture) including OM1 where no kōura were captured. The response variable in these models was the site dissimilarity matrix based on the Bray-Curtis distances between transformed (log x+1) kōura density data. Draftsman plots and correlation matrices were used to identify strongly correlated (r >0.8) variables within the upstream catchment, riparian, instream and physicochemical datasets. Where strong correlations existed, one variable was removed prior to further analysis. Data were also checked for right skewness which can influence DistLM results. None was found.

Using the kōura density resemblance matrix, marginal tests and manual selection tests were performed individually using transformed, normalised upstream catchment, riparian, instream habitat and physicochemical variables. Adjusted R^2 was used as a criterion for variable selection as it takes into account the number of parameters and excludes predictor variables if they add no more to the explained sum of squares. Variables which contributed <3% explanatory power in manual and forward selection models were also excluded. The results of the manual selection process were then compared to models returned using the forward selection and the best-fit processes.

The best-fit process produces only total adjusted R^2 and cumulative R^2 values representing all the variables contained in the model rather than for individual variables, making it difficult to establish if individual variables contribute >3% explanatory power to a particular model. To minimise differences when comparing models, the number of variables in the best-fit models were compared to manual selection results. As a guide, and based on learnings from Chapter 3, best-fit models with the same number of variables as manual selection models or with approximately half the variables included in the original analysis are reported on. Comparisons between manual, forward selection and best-fit models, in conjunction with knowledge of streams and observations of the interaction of habitat variables and kōura ecology, assisted selection of models for upstream catchment, riparian, instream habitat and physicochemical variables that were intuitive, parsimonious and had relatively good predictive power.

Instream habitat was considered the most important reach-scale influence on kōura density. As such, and using the same criteria outlined above, the selected instream habitat model was combined individually with the selected upstream catchment, riparian and physicochemical models to assess the relative effect of each combination and enable further identification of important variables that appeared to influence variation in kōura density between sites. For each combination, manual, forward and best-fit processes were used and the resulting models compared.

The instream habitat model was coupled with the variables identified in the combined process within each of the individual variable groups to establish an overall model that represented the combination of variables which explained the most variation across the entire dataset. Spearman rank correlations were used to examine the strength and direction of association between total kōura density (no. per m^2) and the variables contained in the overall best-fit model. The overall selected best-fit model represented the most logical and parsimonious model accounting for most variation in kōura density (no. per m^2) between sites.

YOY kōura density (no. per m^2, log x+1) dissimilarity matrices based on Bray-Curtis distances were used as the response variable to establish which instream habitat variables (n = 20) explained the most variation between sites where YOY kōura were present. Only sites where YOY (≤ 13 mm OCL) were found were included in this analysis (n = 23). Only the best-fit process was used to establish the overall model and Spearman rank correlations were used to examine the strength and direction of association between YOY kōura density (no. per m^2) and the instream habitat variables contained in the overall best-fit model.

Effects of riparian planting age on kōura

Two-way PERMANOVA (PRIMER 7.0.13) was used to test for overall differences in total kōura and YOY density using riparian planting age group (excluding native sites) and site spatial cluster as factors. Analysis used the dissimilarity matrices based on the Bray-Curtis distances between transformed density (no. per m^2) for total and YOY kōura density. For YOY ($\leq$ 13 mm OCL) kōura analysis, only sites where YOY were found were included in this analysis (n = 20).

Non-parametric Kruskal-Wallis test (STATISTICA v13.3) was used to determine ranked differences in kōura population characteristics among different-age classes of riparian plantings. Riparian sites were classified into five-year age blocks of 1 = pasture (i.e., zero riparian age), 2 = 1-5 years or recent riparian age, 3 = 6-10 years or mid riparian age, 4 = 11-15 years or mid-late riparian age, and 5 = 16+ years or old riparian. Native sites were excluded from this analysis to test the null hypothesis that there would be no difference in kōura population characteristics between riparian-planting age groups at sites formerly or currently grazed.

Non-parametric Mann-Whitney U tests were performed on two combinations of different riparian age class for kōura population variables to establish whether sites with older riparian plantings exhibited kōura population characteristics similar to native forest sites. The paired age classes compared were 11-15 years (mid-late) versus native forest sites and 16+ years (old) versus native forest sites.

4.3 Results

Results of riparian and instream habitat measurements are presented in Chapter 3.3.

4.3.1 Physicochemical characteristics

A summary of physicochemical variables across different site types is shown in Table 4.1. For the physicochemical data for individual sites refer to Appendix 5.

Temperature measurements across all sites ranged from 9.9 to 18.2°C and dissolved oxygen values from 6.7 to 11.1 mg/L. Both mg/L and % saturation dissolved oxygen values were strongly correlated (r_s = 0.72). Specific conductivity ranged between 53.2 and 248.7 µS/cm, pH values ranged from 6.62 to 8.58, turbidity from 0.83 to 14.8 NTU and hardness from 6 to 67 expressed as mg/L of $CaCO_3$. These physicochemical values were used as predictor variables in the DistLM analyses (see below) to evaluate their potential significance relative to upstream catchment, riparian and instream habitat variables, but are not compared between riparian age groups because they were one-off, spot measurements.

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Table 4.1: Mean ± standard deviation (SD) and minimum and maximum values of physicochemical variables by study site type. Pasture n = 3, riparian n = 20, native n = 3.

Variable	**Pasture**				**Riparian**				**Native**			
	Mean	**SD**	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**	**Mean**	**SD**	**Min**	**Max**
Temperature (°C)	13.57	± 0.99	12.44	14.33	14.89	± 1.91	11.10	18.20	11.99	± 2.20	9.95	14.33
Dissolved oxygen (mg/L)	10.71	± 0.24	10.46	10.94	9.75	± 0.97	6.67	11.05	10.80	± 0.40	10.34	11.10
Conductivity (µS/cm @ 25°C)	87.47	± 22.50	69.80	112.80	101.31	± 43.21	57.50	248.70	91.50	± 33.31	53.20	113.80
pH	7.27	± 0.57	6.62	7.69	7.70	± 0.32	7.18	8.58	7.85	± 0.57	7.21	8.30
Turbidity (NTU)	4.99	± 3.53	0.97	7.62	4.14	± 3.70	0.83	14.80	5.32	± 2.35	2.77	7.41
Hardness (as mg/L of $CaCO_3$)	16.67	± 4.16	12.00	20.0	20.90	± 14.15	6.00	67.00	13.67	± 7.09	6.00	20.00

4.3.2 Kōura population characteristics

The total number of kōura captured across 26 study sites totalled 1,143 of which 36% were female, 41% male and 23% unidentified sex. Kōura were found at all sites except OM1. Capture rates between sites with kōura ranged from 1 to 173 individuals for the two electric-fishing passes combined. Across all sites, the smallest kōura captured was 2 mm OCL and the largest 34 mm OCL; the overall mean was 12.25 mm OCL (± SD 3.84). Mature females considered of breeding age (≥17 mm OCL) were found at 19 sites. Only three mature females captured were in berry with either eggs or young still attached and were captured at sites K1, TP1 and TP6 (see Figure 2.1). YOY kōura were found at 23 of the study sites where between two and 128 YOY kōura were caught. Of the kōura whose sex was unable to be determined the majority (78%) were ≤7 mm OCL while the remaining 22% comprised kōura ranging between >7 and 14 mm OCL.

A summary of kōura population characteristics across different site types is shown in Table 4.2. For kōura data for individual sites refer to Appendix 6.

Table 4.2: Mean, standard deviation (SD) and minimum and maximum values for kōura characteristics captured by study site type. Pasture n = 3, riparian n = 20, native n = 3. YOY = young-of-year.

Variable	Pasture				Riparian				Native			
	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Number per m^2	0.31	± 0.20	0.13	0.54	0.62	± 0.74	0.01	2.54	0.45	± 0.53	0.12	1.06
Mean length (OCL mm)	15.56	± 1.70	13.92	17.32	12.72	± 2.93	6.87	19.50	10.06	± 1.93	7.92	11.69
% female	47.75	± 7.62	42.85	56.52	32.68	± 15.87	0	58.30	33.17	± 21.47	15.71	57.14
% male	47.91	± 4.58	43.47	52.63	43.95	± 26.57	0	100	29.05	± 10.72	18.57	40.00
% “unidentified sex”	4.34	± 4.82	0	9.52	23.35	± 26.89	0	82.50	37.78	± 26.00	14.28	65.71
% mature females (≥17 mm OCL)	20.00	± 9.99	10.52	30.43	13.34	± 23.21	0	100	5.87	± 7.47	0	14.28
% YOY (≤13 mm OCL)	38.80	± 12.07	30.43	52.63	54.60	± 25.28	0	85.00	81.75	± 12.16	70.00	94.28

4.3.3 Variables explaining kōura density

For instream habitat, the prevalence of undercut banks, wood and CPOM were not significantly correlated with kōura density, however, the prevalence of root complexes was (r_s = 0.47, p = 0.015). With the exception of percentage gravel substrate (r_s = -0.45, p = 0.024), substrate types and hydraulic conditions showed no significant correlations with kōura density.

DistLM was used to examine which upstream catchment, instream habitat, riparian and physicochemical factors best explained variation in kōura density (no. per m^2) expressed as a resemblance matrix between sites. Results from marginal tests are reported in Appendix 7. Draftsman plots and correlation matrices were used to remove strongly related (r >0.8) variables, thus upstream catchment percentage greywacke was excluded as it was strongly correlated with % volcanic rock (r = -0.91). "Other" geology consisted of small amounts of unconsolidated sands, gravels and alluvium materials with values <10%, so it was also removed from the analysis. For instream variables, CPOM (m^2) and wood (m^2) were highly correlated (r = 0.99) so CPOM was removed from the analysis. No correlations (r >0.8) were present in either the riparian or physicochemical data so all variables were included in those analyses.

Upstream catchment predictors

The manual and forward models each contained seven variables that accounted for 56% of the total variation, while the best-fit model contained eight variables and accounted for 61% of the total variation (Table 4.3).

The best-fit model contained six of the same variables as the manual model but excluded distance to native forest and included percentage sandstone and percentage indigenous vegetation. Five of the variables from forward selection were also in the manual model, but differed from both the best-fit and manual models by including percentage grass vegetation and percentage upstream buffer with woody vegetation.

The final model using the manual selection process based on other model outputs is presented in Table 4.4. It explained a total of 56% (cumulative R^2) of the variation in kōura density among sites using seven upstream catchment variables.

Although total variation accounted for was the same as or less than the forward and best-fit models (Table 4.3), the manual selection model contained one less variable than the best-fit model, provided a robust representation of the major catchment influences observed during GIS analysis (i.e., included gradient and percentage volcanic rock), and was therefore consistent with upstream catchment analysis findings in relation to instream habitat in Chapter 3.

In terms of percentage variation explained by this final model, upstream catchment area accounted for 19%, change in gradient for 8%, percentage upstream scrub vegetation for 10%, followed by upstream catchment geology in volcanic rock at 4% or pre-Taupō ash at 7%, with an additional 3% explained by distance to native forest and 5% by "other" upstream vegetation. Both upstream catchment area and upstream gradient were statistically significant with p values of 0.003 and 0.049, respectively (Table 4.4).

Table 4.3: DistLM analysis summary of the cumulative effect of upstream catchment variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables that led to an increase in adjusted R^2 value and for manual and forward selection processes that contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Catchment area (ha)	1	1	X
Gradient (m/m)	2	-	X
Scrub vegetation (%)	3	2	X
Volcanic rock (%)	4	-	X
Ashes older than Taupō eruption (%)	5	3	X
Distance to native forest (m)	6	4	-
"Other" vegetation (%)	7	5	X
Sandstone (%)	-	-	X
Indigenous vegetation (%)	-	-	X
Grass vegetation (%)	-	6	-
Riparian buffer in woody vegetation (%)	-	7	-
Total number of variables	*7*	*7*	*8*
Cumulative R^2	*0.56*	*0.56*	*0.61*

Table 4.4: Final model selected for the cumulative effect of upstream catchment variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual selection. Significant p values are in bold. Only variables that led to an increase in adjusted R^2 value and contributed >3% to the cumulative R^2 were included in the model.

Variable	**Adjusted R^2**	**SS(trace)**	**Pseudo-F**	**p value**	**Proportion R^2**	**Cumulative R^2**
Catchment area (ha)	0.15	8689.30	5.55	**0.003**	0.19	0.19
Gradient (m/m)	0.21	3780.90	2.57	**0.049**	0.08	0.27
Scrub vegetation (%)	0.28	4649.10	3.51	0.053	0.10	0.37
Volcanic rock (%)	0.30	2039.90	1.58	0.164	0.04	0.41
Ashes older than Taupō eruption (%)	0.35	2900.70	2.40	0.098	0.07	0.48
Distance to native forest (m)	0.35	1499.10	1.26	0.305	0.03	0.51
“Other” vegetation (%)	0.38	2201.40	1.93	0.086	0.05	0.56

Physicochemical predictors

Forward selection and best-fit model results contained the same three variables, temperature, dissolved oxygen and hardness, as the manual selection model which contained four variables and included conductivity. The amount of variation accounted for by the models was very similar and ranged between 49 and 51%. All models are summarised in Table 4.5.

Table 4.5: DistLM analysis summary of the cumulative effect of measured physicochemical variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables which led to an increase in adjusted R^2 value and for manual and forward selection processes that contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Temperature (°C)	1	2	X
Dissolved oxygen (mg/L)	2	1	X
Conductivity (µS/cm @ 25°C)	3	-	-
Hardness (as mg/L of $CaCO_3$)	4	3	X
Total number of variables	*4*	*3*	*3*
Cumulative R^2	*0.51*	*0.49*	*0.50*

The final model using the manual selection process is presented in Table 4.6. Although this model contained one more variable than both the forward selection and best-fit models, it was selected because: i) total variation explained was slightly higher with 1% more than best-fit and 2% more than forward selection; and ii) conductivity was statistically significant, as well as temperature and dissolved oxygen. Temperature and dissolved oxygen collectively explained 42% of the variation, with contributions of 22% and 20%, respectively, while conductivity accounted for 6% and hardness for 3%.

Table 4.6: Final model selected for the cumulative effect of physicochemical variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual selection process. Significant p values are in bold. Only variables that led to an increase in adjusted R^2 and contributed >3% to the cumulative R^2 value were included in the model.

Variable	**Adjusted R^2**	**SS(trace)**	**Pseudo-F**	**p value**	**Proportion R^2**	**Cumulative R^2**
Temperature (°C)	0.19	10171	6.77	**0.002**	0.22	0.22
Dissolved oxygen (mg/L)	0.36	9043.7	7.70	**0.001**	0.20	0.42
Conductivity (µS/cm @ 25°C)	0.41	3057.8	2.81	**0.037**	0.06	0.48
Hardness (as mg/L of $CaCO_3$)	0.42	1368.5	1.27	0.286	0.03	0.51

Riparian predictors

Results from the manual, forward selection and best-fit processes included the same four variables and each model accounted for the same amount of variation (27%). Results for all models are summarised in Table 4.7.

Table 4.7: DistLM analysis summary of the cumulative effect of riparian variables on kōura density (m^2) dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables that led to an increase in adjusted R^2 and for manual and forward selection processes that contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Average stream bank slope (°)	1	1	X
Basal area (m^2 per 20 m^2)	2	2	X
Shrub (%)	3	3	X
Number of saplings (per m^2)	4	4	X
Total number of variables	*4*	*4*	*4*
Cumulative R^2	*0.27*	*0.27*	*0.27*

Since all models contained the same variables and accounted for the same cumulative variation, the manual selection model was chosen for further analysis (Table 4.8). Average stream bank slope accounted for 8% of the variation, basal area and number of saplings 6% each, and shrub cover 7%. No p values were statistically significant.

Table 4.8: Final model selected for the cumulative effect of riparian variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual selection. Only variables which led to an increase in adjusted R^2 and contributed >3% to the cumulative R^2 were included in the model.

Variable	**Adusted R^2**	**SS(trace)**	**Pseudo-F**	**p value**	**Proportion R^2**	**Cumulative R^2**
Average stream bank slope (°)	0.04	3627.50	2.04	0.110	0.08	0.08
Basal area (m^2 per 20 m^2)	0.06	2737.50	1.58	0.182	0.06	0.14
Shrub (%)	0.10	3300.00	1.98	0.122	0.07	0.21
Number of saplings (per m^2)	0.13	2638.00	1.63	0.207	0.06	0.27

Instream habitat predictors

All models contained nine variables with six of the variables in the best-fit model also represented in the manual model (Table 4.9). These two models varied in the types of substrate they represented, and by the inclusion of percentage run habitat which was present in the manual but not in the best-fit or forward models. The forward selection model differed from the manual model by one variable and included thalweg depth in place of percentage run habitat, although variable order differed. The cumulative variation explained was 71% for the manual, 73% for the forward and 74% for the best-fit model, with all models containing nine variables.

Table 4.9: DistLM analysis summary of the cumulative effect of instream variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables that led to an increase in adjusted R^2 and for manual and forward selection processes that contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Run habitat (%)	1	-	-
Macrophyte cover (%)	2	1	X
Root complexes (m^2)	3	2	X
Gravel substrate (%)	4	8	X
Channel width (m)	5	3	X
Cobble substrate (%)	6	4	-
Silt substrate (%)	7	7	X
Boulder substrate (%)	8	9	X
Sand substrate (%)	9	5	-
Thalweg depth (m)	-	6	X
Bedrock substrate (%)	-	-	X
Clay substrate (%)	-	-	X
Total number of variables	*9*	*9*	*9*
Cumulative R^2	*0.71*	*0.73*	*0.74*

The final model using the manual selection process is presented in Table 4.10. Although this model accounted for slightly less variation (R^2 = 0.71) than both the forward and best-fit models (see Table 4.9), the presence of percentage run habitat, percentage cobble substrate and percentage sand substrate were considered better indicators of kōura habitat variation based on field observations than thalweg depth, bedrock and clay substrate which appeared in the other models.

The majority of the variation explained in the final model (Table 4.10) was accounted for by percentage run habitat (15%), macrophyte cover (15%) and root complexes (14%). The remaining variables each contributed between 4 and 6% of the variation. Statistically significant p values occurred for percentage run habitat, percentage macrophyte cover, root complexes (m^2) and percentage boulder substrate.

Table 4.10: Final model selected for the cumulative effect of instream habitat variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual selection. Significant p values are in bold. Only variables that led to an increase in adjusted R^2 and contributed >3% to the cumulative R^2 were included in the model.

Variable	**Adjusted R^2**	**SS(trace)**	**Pseudo-F**	**p value**	**Proportion R^2**	**Cumulative R^2**
Run habitat (%)	0.11	6789.70	4.13	**0.011**	0.15	0.15
Macrophyte cover (%)	0.24	7312.30	5.23	**0.003**	0.15	0.30
Root complexes (m^2)	0.37	6389.30	5.46	**0.001**	0.14	0.44
Gravel substrate (%)	0.39	1913.30	1.69	0.166	0.04	0.48
Channel width (m)	0.42	2385.30	2.22	0.078	0.05	0.54
Cobble substrate (%)	0.44	1828.00	1.77	0.143	0.04	0.58
Silt substrate (%)	0.46	1618.70	1.62	0.183	0.04	0.61
Boulder substrate (%)	0.52	2872.00	3.23	**0.017**	0.06	0.67
Sand substrate (%)	0.54	1665.30	1.98	0.107	0.04	0.71

Combined instream habitat and physicochemical predictors

Using the final model outputs from manual selection for physicochemical variables (Table 4.6) and instream habitat (Table 4.10), a combined model was analysed using manual, forward and best-fit selection processes to ascertain if adding physicochemical variables to instream habitat variables would improve model prediction (Table 4.11). In total, 13 variables were considered. The addition of spot physicochemical variables to instream habitat variables improved the model variance explained from 71% to 77% for manual selection and from 74% to 78% for best-fit, but weakened the model for forward selection reducing variation explained from 73% to 71%. Forward selection only produced a model with seven variables as the procedure stops when there is no further possible improvement to the selection criterion (Table 4.11).

Dissolved oxygen appeared in all models and hardness in two, so these two physicochemical parameters were selected to include in the pool of variables for development of an overall model.

Table 4.11: DistLM analysis summary of the cumulative effect of instream habitat and physicochemical variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables that led to an increase in adjusted R^2 for manual and forward selection processes and contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Run habitat (%)	1	-	-
Macrophyte cover (%)	2	1	X
Root complexes (m^2)	3	2	-
Gravel substrate (%)	4	-	X
Channel width (m)	5	5	X
Cobble substrate (%)	6	6	X
Silt substrate (%)	7	7	X
Boulder substrate (%)	8	-	X
Dissolved oxygen (mg/L)	9	4	X
Hardness (as mg/L of $CaCO_3$)	10	3	-
Sand substrate (%)	-	-	X
Temperature (°C)	-	-	X
Conductivity (µS/cm @ 25°C)	-	-	X
Total number of variables	*10*	*7*	*10*
Cumulative R^2	*0.77*	*0.71*	*0.78*

Combined instream habitat and riparian predictors

Using the model outputs from final manual selection for riparian and instream habitat variables (Table 4.8 and Table 4.10), the combined model was run using manual, forward and best-fit selection processes to ascertain if adding riparian variables to instream habitat variables would improve model prediction (Table 4.12).

This combination improved the model from 71% to 75% variance explained using manual selection, 73% to 77% for forward selection and 74% to 77% for best-fit.

Number of saplings and average stream bank slope appeared in both the manual and best-fit models, so these variables were included in the pool of variables for development of an overall model.

Table 4.12: DistLM analysis summary of the cumulative effect of instream habitat and riparian variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual, forward and best selection processes. Only variables that led to an increase in adjusted R^2 and for manual and forward selection processes and contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Run habitat (%)	1	-	-
Macrophyte cover (%)	2	1	X
Root complexes (m^2)	3	2	X
Gravel substrate (%)	4	-	X
Channel width (m)	5	3	X
Cobble substrate (%)	6	4	X
Silt substrate (%)	7	-	-
Boulder substrate (%)	8	-	X
Average stream bank slope (°)	9	5	X
Number of saplings (per m^2)	10	-	X
Shrub (%)	-	6	-
Sand substrate (%)	-	-	X
Total number of variables	*10*	*6*	*9*
Cumulative R^2	*0.75*	*0.77*	*0.77*

Combined instream habitat and upstream catchment predictors

Using the model outputs from manual selection for instream habitat and upstream catchment variables (Table 4.10 and Table 4.4), a combined model was analysed using manual, forward and best-fit selection processes to ascertain if adding upstream catchment variables to instream habitat would improve model prediction.

This combination improved model variance explained from 71% to 75% using manual selection with a total of ten variables, from 73% to 79% with nine variables using forward selection, and from 74% to 79% for best-fit with nine variables (Table 4.13).

Upstream catchment area appeared in all three models while “other” vegetation and pre-Taupō ashes appeared in the best-fit and forward selection models. All three variables were included in the pool of variables for development of an overall model.

Table 4.13: DistLM analysis summary of the cumulative effect of instream habitat and upstream catchment variables on kōura density (no. per m^2) dissimilarity among 26 study sites using manual, forward and best-fit selection processes. Only variables which led to an increase in adjusted R^2 and for manual and forward selection processes and contributed >3% to the cumulative R^2 were included in the model. Numbers = rank of variable in model; X = presence of variable in model.

Variable	**Manual**	**Forward**	**Best-fit**
Run habitat (%)	1	-	-
Macrophyte cover (%)	2	1	X
Root complexes (m^2)	3	3	X
Gravel substrate (%)	4	-	-
Catchment area (ha)	5	2	X
Channel width (m)	6	8	X
Cobble substrate (%)	7	5	X
Silt substrate (%)	8	-	-
Boulder substrate (%)	9	-	-
Sand substrate (%)	10	6	X
“Other” vegetation (%)	-	7	X
Ashes older than Taupō eruption (%)	-	4	X
Scrub vegetation (%)	-	9	-
Total number of variables	*10*	*9*	*9*
Cumulative R^2	*0.75*	*0.79*	*0.79*

Overall model for combined instream habitat, physicochemical, riparian and upstream catchment predictors

The selected model for instream habitat (Table 4.10) was combined with variables identified by analysing the individual combinations of instream habitat with physicochemical, riparian and upstream catchment variables (Tables 4.11, 4.12 and 4.13). A total of 16 variables were combined for the overall model (summarised in Appendix 8) distilled from an original selection of 50 variables.

A best-fit process was used for development of the overall model. It was considered the most robust process to use given the number of variables as well as the inability to establish individual R^2 values for variables identified using the best-fit process in previous models. The selected overall best-fit model explained a total of 83% of the total variation in the dissimilarity of total kōura densities between sites by using ten variables from the subset of 16 variables refined through the analysis process. These variables comprised four instream habitat, two riparian, one physicochemical and three upstream catchment variables (Table 4.14). This combined model is the most logical and parsimonious, explaining substantial variation in kōura density between study sites.

Correlations for the selected variables in the best-fit model with total kōura density to identify the direction of effect revealed four of the variables had statistically significant individual correlations (Table 4.14). The overall best-fit model explained 83% of variation in total kōura density between sites and contained a total of 10 variables: three upstream catchment variables (catchment area (-), percentage “other” vegetation (-) and pre-Taupo ash geology (-); two riparian variables (number of saplings (-) and stream bank slope (+)); one physicochemical variable (dissolved oxygen (+); and four instream habitat variables related to substrate (percentage cobble (+) or sand (+)); and organic cover (cover by macrophytes (-) or root complexes (+)).

Table 4.14: Spearman rank correlations and associated p values for each variable in the overall final model with total kōura density are presented. Significant p values for correlations are in bold.

Variable contained in the overall total kōura density model	**r_s value**	**p value**
Macrophyte cover (%)	-0.69	**0.001**
Root complexes (m^2)	+0.31	0.118
Cobble substrate (%)	+0.15	0.478
Sand substrate (%)	+0.05	0.801
Number of saplings (per m^2)	-0.15	0.468
Dissolved oxygen (mg/L)	+0.26	0.232
Catchment area (ha)	-0.53	**0.004**
“Other” vegetation (%)	-0.51	**0.007**
Ashes older than Taupō eruption (%)	-0.28	0.159
Average stream bank slope (°)	+0.39	**0.050**

4.3.4 Variables explaining young-of-the-year kōura density

All instream habitat variables (n = 20) were tested using DistLM to determine which variables were influencing YOY dissimilarity between sites. Only sites with YOY present (n = 23) were included in the analysis and the best-fit model process was used. Marginal tests are presented in Appendix 9.

The best-fit model with the same number of variables as the overall total kōura density model (n = 10) was selected for consistency and is summarised in Table 4.15. The selected best-fit model contained six of the same ten variables present in the model selected to best represent instream habitat and influences on total kōura density dissimilarities (Table 4.10). The YOY model did not contain percentage boulder and silt substrates or channel width which were replaced by percentage clay substrate, percentage “other” habitat, thalweg depth and reach gradient.

Correlations for the selected instream habitat variables in the best-fit model with YOY kōura density to identify the direction of effect revealed three of the variables had statistically significant individual correlations (Table 4.15). The overall best-fit model explained 80% of variation in YOY kōura density between sites and contained a total of ten variables: four variables related to substrate type (percentage gravel (-), cobble (+), sand (+) or clay (-)); four variables related to hydraulic conditions (percentage run habitat (-), “other” habitat (+), thalweg depth (-) and reach gradient (+)); and two variables representing organic cover (macrophyte cover (-) and root complexes (+)).

Table 4.15: Spearman rank correlations and associated p values for each variable in the overall final model with total kōura density are presented. Underlined variables were also selected in the habitat model for total kōura density. Significant p values for correlations are in bold.

Variable contained in the overall YOY kōura density model	**r_s value**	**p value**
<u>Macrophyte cover (%)</u>	-0.63	**0.001**
<u>Gravel substrate (%)</u>	-0.52	**0.011**
<u>Cobble substrate (%)</u>	+0.01	0.946
<u>Sand substrate (%)</u>	+0.07	0.744
Clay substrate (%)	-0.19	0.393
<u>Root complexes (m^2)</u>	+0.55	**0.006**
Thalweg depth (m)	-0.27	0.202
<u>Run habitat (%)</u>	-0.27	0.218
“Other” habitat (%)	+0.12	0.598
Gradient (reach) (m/m)	+0.21	0.332

4.3.5 Effects of riparian planting age on kōura

Spearman rank correlations were used to examine the strength and direction of association between riparian planting age (excluding native) and total kōura and YOY density (both no. per m^2) and mean kōura length (mm). No statistically significant correlations were revealed; total kōura density $r_s = 0.009$, YOY density $r_s = 0.120$ and average length $r_s = -0.215$.

PERMANOVA revealed no significant effect of riparian age group (Pseudo-$F_{4, 22} = 1.339$, p = 0.277) or site spatial cluster (Pseudo-$F_{2, 22} = 0.771$, p = 0.597) on total kōura density (no. per m^2) dissimilarity among sites. There was also no significant interaction between these two factors (Pseudo-$F_{4, 22} = 0.969$, p = 0.475). For YOY kōura density (no. per m^2) dissimilarity among sites with YOY kōura present, no significant effect was evident for site spatial cluster (Pseudo-$F_{2, 19} = 0.938$, p = 0.484) or riparian age group (Pseudo-$F_{4, 19} = 1.219$, p = 0.331) and there was no significant interaction between these two factors (Pseudo-$F_{3, 19} = 1.207$, p = 0.316).

Kruskal-Wallis tests were undertaken to further investigate influences of different riparian age groups on the full range of kōura population characteristics. No significant differences between groups were detected (Table 4.16). For completeness, box plots for total kōura density, mean length (OCL mm) and YOY density are presented in Figures 4.1, 4.2 and 4.3. Patterns related to riparian age group were not visually apparent for total or YOY kōura densities, but mean length did appear to decline and variability increase from sites in pasture to those in mid-late riparian planting, before increasing again in old riparian plantings (Figure 4.2). The high variability in later stage riparian planting presumably contributed to the lack of a statistically significant effect.

Mann-Whitney U tests for differences between mid-late and old riparian age groups and native forest sites for kōura population characteristics also revealed no significant differences (Table 4.17).

Table 4.16: Results of Kruskal-Wallis analysis for kōura population characteristics between riparian planting age groups: 0 years = pasture, 1-5 years old (recent), 6-10 years old (mid), 11-15 years old (mid-late) and 16+ years old (old). No p values were statistically significant (<0.05) (n = 23).

Variable	**H value**	**p value**
Total (no. per m^2)	3.08	0.54
Female (no. per m^2)	0.92	0.92
Male (no. per m^2)	2.12	0.71
"Unidentified sex" (no. per m^2)	2.62	0.62
Mature female (no. per m^2)	3.28	0.51
YOY (no. per m^2)	3.46	0.48
Mean length (mm)	5.44	0.24
Minimum length (mm)	7.90	0.09
Maximum length (mm)	4.01	0.40
Median length (mm)	3.09	0.54

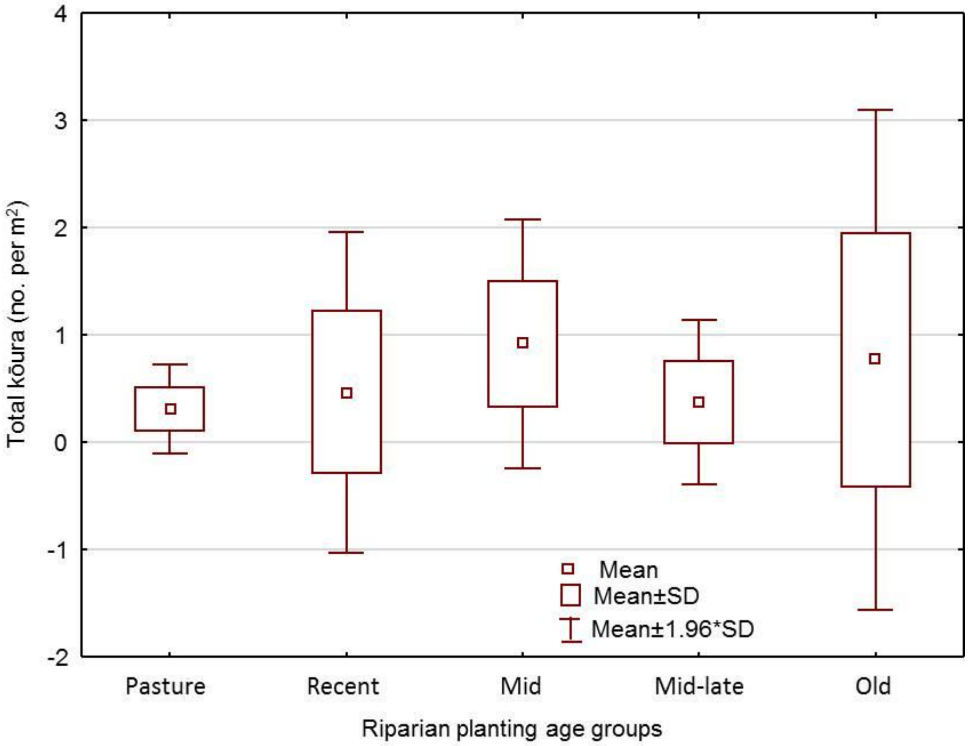

Figure 4.1: Box plot of kōura density (no. per m^2) by riparian planting age group for 23 Pirongia, Whaingaroa and Whatawhata stream sites. Pasture (n = 3), Recent (n = 8), Mid (n = 3), Mid-late (n = 5), Old (n = 4).

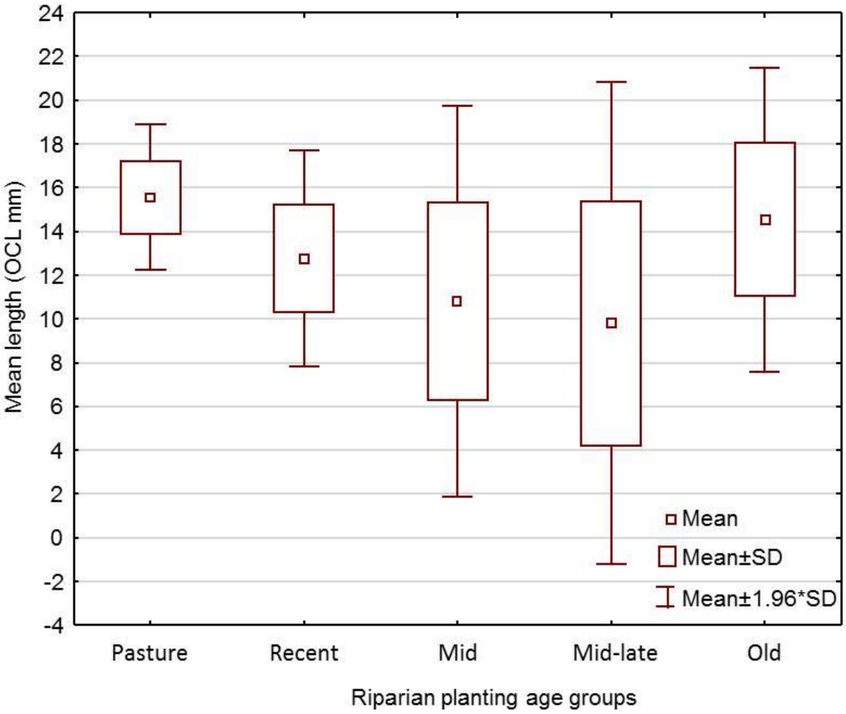

Figure 4.2: Box plot of mean length (OCL mm) of kōura by riparian planting age group for 23 Pirongia, Whaingaroa and Whatawhata stream sites. Pasture (n = 3), Recent (n = 8), Mid (n = 3), Mid-late (n = 5), Old (n = 4).

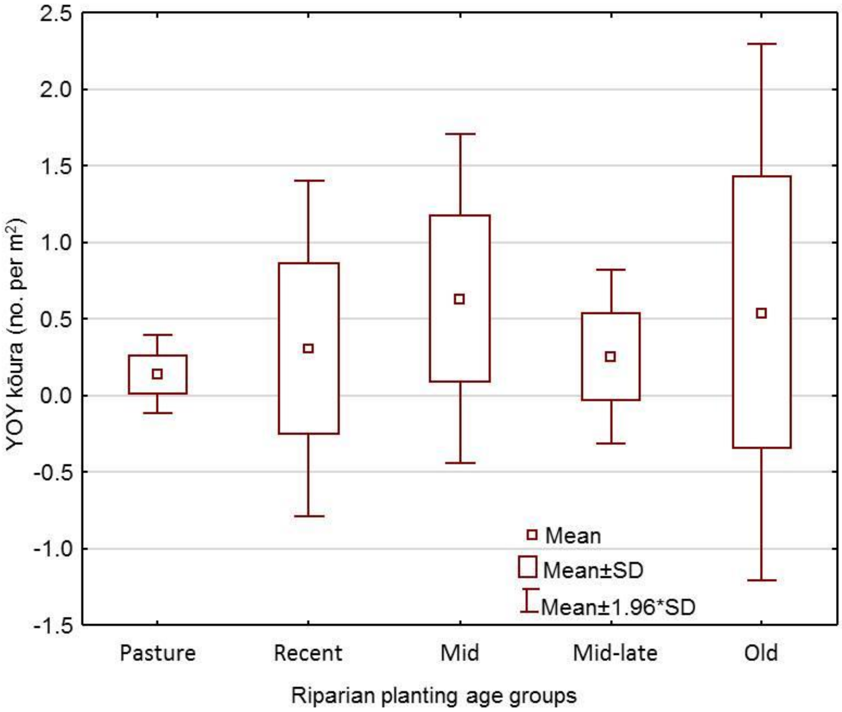

Figure 4.3: Box plot of YOY kōura density (no. per m^2) by riparian planting age group for 20 Pirongia, Whaingaroa and Whatawhata stream sites. Pasture (n = 3), Recent (n = 7), Mid (n = 3), Mid-late (n = 4), Old (n = 3).

Table 4.17: Results of Mann-Whitney U test for kōura population characteristics between 11-15 years old planting sites (mid-late) and native forest sites and 16+ years old planting sites (old) and native forest sites. Mid-late riparian sites n = 5, old riparian sites n = 4, native sites n = 3. No p values were statistically significant (<0.05) (n = 23).

Variable	**11-15 years (Mid-late) vs Native**		**16+ years (Old) vs Native**	
	U statistic	**p value**	**U statistic**	**p value**
Total (no. per m^2)	3.00	1.00	5.00	0.86
Female (no. per m^2)	6.00	0.77	3.00	0.86
Male (no. per m^2)	6.00	0.77	3.50	0.86
“Unidentified sex” (no. per m^2)	7.00	0.23	5.00	0.38
Mature female (no. per m^2)	6.00	0.77	6.00	0.48
YOY (no. per m^2)	5.00	1.00	1.00	0.86
Mean length (mm)	5.00	0.55	1.00	0.11
Minimum length (mm)	7.00	0.46	5.00	0.86
Maximum length (mm)	6.00	0.37	5.00	0.22

4.4 Discussion

The focus of this chapter has been on: i) determining the relative influence of upstream catchment, physicochemical and/or reach-scale instream habitat and riparian factors on kōura density and size between small stream sites in the western Waikato; and ii) determining the effects of riparian planting age on kōura density and size compared to pastoral and native forest sites.

All riparian planting sites were fenced, and along with the native forest reference sites were not subject to stock grazing. Pasture sites were unfenced and accessible to stock, and were included to represent conditions that prevailed prior to riparian fencing and planting. Sites were clustered across three areas, each with native forest and pasture sites represented to ensure spatial bias was reduced, as confirmed in the PERMANOVA analyses. As such, the results would be broadly applicable to other wadeable, hill-country North Island streams. Associations between upstream catchment and riparian characteristics on instream habitat are explored in detail in Chapter 3, and association of fish communities on kōura are assessed in Chapter 5.

4.4.1 Kōura populations in study streams

Sites were selected because of the confirmed or likely presence of kōura. With the exception of OM1 (see Figure 2.1), kōura were found at all sites, although density varied widely from 0.01 to 2.5 kōura per m^2. The absence of kōura at OM1 was unexpected as habitat appeared to be highly suitable with the only apparent difference being conductivity (248.7 μ/S cm @ 25°C), which was higher than all other sites and substantially higher than the low conductivity of 50 μS/cm indicated in Death *et al.* (2017) to represent favourable conditions for kōura. The proximity of the site to the sea (1.14 km), coupled with being <3 m above sea level, indicates a saline influence may have been responsible for the lack of kōura. Interestingly, adult Galaxiidae species were found in the same sample reach, however, their salinity tolerance is likely higher given their diadromous lifecycle, and greater mobility relative to kōura.

The association of kōura with stream banks, instream cover and slow flowing water is well known (Parkyn & Collier, 2004; Jowett *et al.*, 2008), and was confirmed in the present study during field observations. Although all available instream habitat was sampled at every study site, the majority of kōura were generally found within 0.5 m the bank or in pools. Kōura were very rarely collected from the faster flowing mid-sections of the larger streams unless an obstacle provided cover and an area of decreased velocity downstream. Kōura are primarily nocturnal and generally seek cover during daylight (Collier *et al.*, 1997) which is when sampling was undertaken, influencing their association with particular habitat types during capture.

All physicochemical measurements were one-off samples that only characterise a discrete point in time. While one-off measurements are not useful to characterise physicochemical characteristics of particular sites, particularly those that vary diurnally such as temperature, dissolved oxygen and pH, most of my readings were taken around the same time of day (early morning) and so provide a consistent measure to compare among sites. In the present study I have used the physicochemical measurements to help account for variation in density of kōura among sites that was not accounted for by instream habitat but they are not used to interpret potential mechanisms affecting kōura.

In general, study sites exhibited "soft" water characteristics and had $CaCO_3$ mg/L values ≤59 $CaCO_3$ mg/L (ANZECC, 2000), with all calcium values well above the minimum concentration of ≥5 mg/L of calcium recommended for *Paranephrops* aquaculture (Hollows, 2016). Dissolved oxygen was generally consistent across sites, however, one site (TM1; see Figure 2.1) was lower at 6.7 mg/L, but this is well above the 48 hour LC_{50} of 0.77 mg/L (±0.06 mg/L) established for kōura by Landman *et al.* (2005) at 15°C. Water temperatures during the early morning were all <18.2 °C, well below the five day thermal tolerance of 26.2°C established for kōura in Albert *et al.* (2015). All pH values were consistent with those established for kōura by Olsson *et al.* (2006) of between 4.1 and 7.9. Dissolved oxygen was the only physicochemical variable included in the final models for total and YOY kōura density, presumably accounting for the low densities and slightly lower oxygen concentration coinciding at TM1.

4.4.2 Factors affecting kōura

In Missouri/Arakansas, USA, both Nolen *et al.* (2014) and Magoulick *et al.* (2017) found catchment-scale variables more important in explaining crayfish presence, while reach-scale variables were more influential on crayfish abundance.

Given that all sites in the present study were pre-selected for the presence or expected presence of kōura, findings from the current study were consistent with those studies, with 71% of variation in total kōura density between study sites explained by instream habitat, compared to 56% explained by upstream catchment predictors and 27% explained by riparian predictors. Physicochemical predictors did account for 51% variation in kōura density, however, as noted above, these were one-off measurements and only one variable appeared in the final model. It is likely that one-off physicochemical measurements are more representative of catchment-scale factors rather than reach-scale variables.

For upstream catchment, the two most important predictor variables were catchment area and gradient, followed by various classes of geology, upstream land-cover and distance to contiguous native forest. Stream discharge and channel width generally increase with increasing catchment area (Pearson, 1992), and both factors have previously been identified as exerting negative influences on kōura abundance and distribution (Jowett *et al.*, 2008) which was confirmed in this study with upstream catchment size significantly negatively associated with total kōura density. Channel width also featured in models for instream habitat, confirming the link between greater channel width and increasing catchment size. Indeed, Hicks (2003) also found catchment size negatively influenced kōura density in some of the same Waikato hill-country streams sampled in this study, with density differences more closely associated with the amount of available edge habitat.

Upstream gradient and catchment geology, both interact to influence channel morphology, water velocity and substrate composition; all factors related to crayfish presence and abundance (Magoulick *et al.*, 2017). Using probability methods, Dyer *et al.* (2013) found geology and the associated soil types were the most consistent predictors of *Orconectes* spp. occurrence in an Oklahoma/Arkansas stream (USA). In the present study, geology had strong explanatory power when examined in relation to instream habitat dissimilarities (Chapter 3), and again featured in explanation of dissimilarities in kōura density between sites in the final model for both total and YOY kōura density.

The distinct geology types present amongst the Waikato study sites likely influenced substrate type, and this was apparent in explaining the dissimilarities in kōura density through the significance of boulder, cobble, gravel and sand/silt substrates. Kōura preferences for substrate have previously been established in Olsson *et al.* (2006) and Jowett *et al.* (2008) who found adult kōura preferred cobbles rather than sand and boulders while juveniles were often found associated with finer substrates and near the stream edge. A prevalence of boulders generally indicates steep catchments and faster hydraulic conditions whereas gravel, sand and silt are often indicative of slower flowing streams with lower channel gradient and more run and pool habitat. In this study, both percentage cobbles and percentage sand featured in the overall model for explaining kōura density dissimilarities between sites, however, both were weakly positively correlated with total kōura density. Field observations indicated a variety of habitats in many of the steeper streams, even when they were dominated by boulders, including low velocity zones with cobbles, root mats and detritus where the majority of kōura were captured. Conversely, several other streams had areas of soft sand and silt which coincided with high densities of kōura, particularly when streams were small, located in the stream headwaters, and where there were pool and root complex habitat available. Indeed, substrate variety was identified as an important predictor of positive kōura abundance across Waikato streams by Death *et al.* (2017) using Boosted Regression Tree (BRT) modelling.

Some of the sampling sites dominated by smaller substrates and runs exhibited evidence of large flow variability driven by larger upstream catchments dominated by pasture and scrub. These streams appeared to be unsuitable for kōura persistence, a conclusion that generally aligns with the findings of Parkyn and Collier (2004) and Jowett *et al.* (2008) who found, respectively, that native forest but not pasture provided stable instream refugia during flooding and that kōura preferred areas with slower water velocities. Using BRT modelling, Death *et al.* (2017) also identified kōura abundance was positively associated with small floods in the Waikato, further confirming the influence of both flow and refugia on kōura populations.

Distance to native forest also featured as a predictor variable in the analysis of kōura density and upstream catchment variables, although it did not feature in the final overall model. This predictor may be important for kōura for several reasons including: i) provision of vegetative inputs such as CPOM and wood are likely to be higher closer to native forest (Richards & Host, 1994); ii) the role of catchment vegetation in mitigating overland flow and reducing flow variability (Richards & Host, 1994; Ministry for the Environment, 2001); iii) native forest may provide stable refugia areas for kōura populations and contribute to dispersal downstream after high flow events (Sponseller *et al.*, 2001; Parkyn & Collier, 2004); and iv) study sites located close to native forest are generally in the upper reaches of catchments which is often characteristic of better quality habitat suitable for kōura as noted above. Land-cover, particularly associated with the riparian zone, has previously been established as an important predictor of crayfish abundance in overseas studies, such as Burskey and Simon (2010) who found the presence of riparian forest was a significant positive predictor of four crayfish genera (*Cambarus, Fallicambarus, Orconectes* and *Procambarus*) in an Indiana (USA) catchment.

The presence of "other" vegetation in the overall total kōura density model is primarily indicative of the presence of exotic forest in the upstream catchments of many of the Whatawhata spatial cluster sites which often coincided with low numbers of kōura as confirmed by "other" vegetation being significantly negatively correlated with kōura density in this study. A previous study by Hicks and McCaughan (1997) found no significant differences in kōura densities between pasture, exotic forest and native forest sites in Waikato hill-country streams indicating that the results from this study may reflect the unsuitable flow variability, dominance of softer, finer substrates and lack of instream habitat suitable for kōura persistence.

Related to substrate and hydraulic conditions in the kōura models were cover of macrophytes which featured strongly in the instream model, accounting for 15% variation. Kōura are known to use macrophytes both for cover and as a food source, particularly in pasture (Parkyn, 2000), however, macrophyte cover was limited in extent in this study primarily due to many sites having unsuitable substrate and hydraulic conditions for their establishment, or being heavily shaded by riparian vegetation. At the nine sites where they were present, streams had lower gradients, more run habitat, softer substrates and relatively little shading, all of which provide suitable conditions for macrophytes to establish. Six of the nine sites which contained macrophytes had very low densities of kōura. Indeed, the sites which contained the greatest instream macrophyte cover had some of the lowest kōura densities, findings which were supported by the statistically significant negative correlation of macrophytes with both total and YOY kōura densities. However, these results were probably confounded with high flow variability and a general lack of indigenous forest in the upstream catchments of these sites which may have provided unsuitable habitat for kōura persistence.

Interestingly, riparian factors accounted for only 27% of the variation in total kōura density between sites. This was similar to variation explained by riparian variables on instream habitat dissimilarities in Chapter 3. Stream bank slope featured as contributing the most variation, however, as previously discussed in Chapter 3, this variable is more closely related to other catchment variables such as geology, catchment size, topography and stream gradient rather than directly influenced by riparian condition. It was expected that stream bank slope would have been associated with increased stream gradient and water velocity and exerted a negative influence on kōura density, however, it was significantly positively correlated to total kōura density. This positive correlation reflects finding in this study that abundant kōura were often found in smaller streams located in incised gullies with many at higher elevation in steeper headwater catchments, all characteristics leading to increased stream bank slope and gradient.

Other influential riparian factors were all associated with factors expected to vary with riparian planting ages, such as basal area, shrub cover and number of saplings, all of which could influence preferred kōura habitat through provision of wood, CPOM and root complexes that provide refugia from floods and predation (Usio & Townsend, 2000; Parkyn & Collier, 2004).

Indeed, the prevalence of root complexes as an important instream habitat predictor for kōura density differences between sites was evident throughout these analyses, although it was only weakly correlated to total kōura density. However, field observations confirmed strong associations of kōura with these habitat features, particularly in areas where velocity was low. In contrast, root complexes did not appear to be influential in providing habitat in fast flowing runs or riffle areas, again suggesting velocity is an important governing factor for kōura habitat selection. This finding aligns with those of Jowett *et al.* (2008) and Burskey and Simon (2010) who found that a riffle/run "score" was a negative predictor of crayfish in their Indiana (USA) study. Interestingly, the site with the most root complexes in the present study, (TP1; see Figure 2.1) did not have the highest number of kōura. The root complexes at TP1 were extensive and in places covered the majority of the channel, however, many were located in fast flowing run sections which were obviously not habitat preferred by kōura.

Parkyn (2000) found that tree roots, where they were available, were the most commonly used kōura habitat in a pasture stream, highlighting the link between root complexes and remnant riparian trees. When found in suitable velocity regimes, tree ferns (*Dicksonia* spp. and *Cynatea* spp) and mahoe (*Melicytus ramiflorus)* appeared to be the best providers of root complexes, with older established remnant vegetation providing more extensive and complex root habitat along banks.

Root complexes can also indicate greater stream bank stability, however, how well they assist in achieving this can be dependent on a combination of factors including: i) stream bank height relative to root depth penetration; ii) bank slope; iii) flow variability; iv) bank material composition and its susceptibility to erosion; and v) whether there are other features present such as bedrock or boulders which can provide some bank stability and deflect flow (Reeves *et al.*, 2004) All these factors can influence the effectiveness of riparian planting in providing submerged root habitat, as well as other key kōura habitats such as undercut frequency/sizes and overhanging vegetation cover. These conclusions are in line with field observations that the ability of riparian planting to stabilise banks and provide optimal kōura habitat was limited where stream channels were incised, banks comprised erodible material and streams drained larger catchments which likely resulted in higher flow variability. In the present study, such sites yielded low kōura numbers, however, these influences may also have been confounded by a combination of other influences not evident at a reach-scale.

These findings partially support Hypothesis 1 (see section 4.1) in which a positive response to kōura density was expected with increased root complexes, wood, CPOM and undercut banks, although only root complexes were statistically significant in correlation analyses. Parkyn (2000) also found root complexes to be important habitat for kōura in both pasture and native sites in Waikato hill-country streams, and noted the use of CPOM and wood by kōura was limited in native streams even when abundant. Additionally, in this study, the expectation that kōura density would decline in response to increased hydraulic stress and larger substrates was not evident at the reach-scale, although the statistically significant negative association between total kōura density and percentage gravel substrate may indicate the unsuitability of gravel dominated sites as kōura habitat due to the mobile nature of gravel and its inability to provide interstitial space for kōura refugia and cover compared to larger substrates such as cobbles.

The YOY model is supported by previous findings in Jowett *et al.* (2008) that smaller kōura prefer finer substrates characteristic of slower flowing areas such as gravel and sand. Indeed, the majority of very small kōura (<5 mm OCL) were captured in these habitat types during field sampling. The presence of "other" habitat (dominated by cascade sequences) and reach gradient in the model would likely suggest these two factors would negatively impact YOY kōura due to an increase in water velocity, however, when tested for correlation both were weakly positively correlated to YOY kōura density. Sites did vary over the 50 m sample reaches and, although gradient may have been high and cascade sequences present, most sites had a variety of habitat which would have provided slower flowing areas for YOY kōura. The appearance of thalweg depth in the YOY model may reflect the wide range of both shallow and deeper water where YOY were captured in this study. Thalweg depth was weakly negatively correlated with YOY kōura in this study, supporting, at least in part, previous findings of Jowett *et al.* (2008) who found most YOY associated with very shallow water. Clay substrate featured in the YOY model and was weakly negatively correlated to YOY kōura density. Clay substrate was relatively rare amongst study sites and was only found in any quantity at three sites which coincided with YOY presence (TP2, TP5 and TU2; see Figure 2.1). All three sites had sections of hard clay substrate associated with areas of higher velocity which had scoured and removed other overlying substrates. Despite YOY being present in these sample reaches, they were not captured in these clay substrate areas and at all three sites were associated with other suitable habitats such as soft sediment and slow flowing areas dominated by smaller substrates. As with the total kōura model, macrophyte cover featured in the YOY model and was significantly negatively correlated with YOY kōura, and likely reflected the low number of sites with macrophyte cover present coupled with the low density of YOY kōura at these sites.

In summary, instream habitat appeared to be the main driver of variation in total and YOY kōura densities among the small Waikato streams sampled in the present study. As reported in Chapter 3, upstream catchment-scale factors are key drivers on instream habitat characteristics. Accordingly, the analyses conducted in this chapter support Hypothesis 2, (see section 4.1) with differences in kōura density among sites influenced more by key aspects of instream habitat, such as substrate size and hydraulic habitat that are regulated by catchment-scale variables rather than local-scale riparian planting. Nevertheless, reach-scale riparian planting can interact with these larger scale drivers to influence local-scale habitat features such as root mats and bank undercuts that are commonly used by kōura.

4.4.3 Effects of riparian planting age on kōura

This study used a space-for-time approach to investigate the effects of riparian planting age on kōura populations, by selecting similar sites (small streams draining western Waikato hill-country) that were fenced and planted with riparian vegetation at different times. Although all planted sites were formerly in pasture they had not experienced the same antecedent conditions by virtue of being planted at different times. The modelling approach discussed above accounted for various aspects of riparian plant growth (e.g., basal area, no. of saplings), but it did not explicitly investigate the effects of riparian planting age. The riparian variables collected were surrogate measurements based on the premise that those characteristics would change as plantings matured and sites would progress to be more similar to native forest over time. However, the type of species used in planting, planting density, growing conditions, the extent of existing remnant native vegetation and/or the presence of individual large trees at a study site would all influence riparian vegetation measurements at a reach-scale irrespective of planting age. Even vegetation at native sites varied in composition and structure and was likely influenced by differences in site elevation, soil type, aspect, forest age, elevation and proximity to the sea and its mitigating influence on climate. Additionally, the influence of stock grazing on riparian vegetation and instream habitat at pasture sites was a function of stream bank slope governing stock accessibility, bank substrate and stability, and the presence of occasional large trees or remnant vegetation in steep areas.

Also, a range of other factors could potentially affect the abundance of kōura at the reach-scale as revealed by the modelling approach used in the present study. Findings have revealed strong influences from upstream catchment variables such as catchment size, geology and vegetation along with their subsequent influence on variables measured at the reach-scale such as stream bank slope and stream substrate composition.

The hypothesis that kōura abundance, including YOY kōura, would be greater in older riparian aged planting sites than in pastoral or early riparian planting sites (Hypothesis 3 in part; see section 4.1) was not supported in the present study. Rather, riparian planting age appeared to be exerting secondary influences on kōura compared to the combined influence of upstream catchment and instream habitat variables. Additionally, PERMANOVA results for both total kōura density and YOY density dissimilarity between sites for riparian age group or site spatial clusters showed no statistically significance. Establishing differences in kōura densities even between different land-uses have previously been attempted with mixed results. Hicks and McCaughan (1997) found no statistically significant differences in kōura densities between Waikato hill-country streams in pasture, exotic forestry or native forest. However, in the same general location, Parkyn *et al.* (2002) found kōura densities were generally greater in native forest streams than in pasture streams, although, as in the present study, densities varied greatly among study streams and between land-cover.

Confounding the ability to establish any strong link between riparian planting age and kōura density is the large variability in kōura capture rates between sites of similar ages, particularly in the recent (0.018-2.274 m^2), mid-late (0-1.037 m^2) and old (0.012-2.54 m^2) categories. As established in Chapter 3, the influences on instream habitat driven by catchment-scale processes are likely contributing to some of these differences. However, the absence or low densities of kōura at a particular site may be due to a range of factors not quantified in this study, or acting synergistically to influence kōura density. Nevertheless, there is no doubt that riparian planting age had some influence on instream habitat based on both field observations and findings in Chapter 3, such as increased shade with planting age. The similarity between riparian planting sites older than 11 years and native sites for instream habitat established in Chapter 3 was also reflected in kōura densities which did not differ significantly, supporting the second part of Hypothesis 3, suggesting that older riparian planting sites providing similar instream conditions for kōura as native sites.

Chapter 5

Association of fish communities with kōura distribution and density

5.1 Introduction

The most important influence on native fish community composition in New Zealand is diadromy which can regulate abundance of migratory species through a combination of large-scale variables such as site elevation and downstream channel gradient, distance downstream from the sea and the presence or absence of any barriers preventing upstream migration (Hanchet, 1990; McDowall, 1990; McDowall, 1993; Joy & Death, 2001). Additionally, New Zealand fish community composition is influenced by individual species' swimming and climbing abilities and their instinctive drive to migrate (McDowall, 1993). McDowall (1996) described fish communities dominated by diadromous species as 'open communities' with their structure driven by the influx of species from outside the community (i.e., the sea), and contrasted these to 'closed communities' structured by biotic and abiotic influences in the immediate vicinity.

Consequently, the ability to establish links between species diversity and factors such as land-use, habitat and/or biotic influences can be confounded due to the over-riding effect of diadromy (Hanchet, 1990; Jowett & Richardson, 1996). For example, in the Grey River, South Island, Jowett *et al.* (1996) found it difficult to separate the influence of land-use, diadromy and physical habitat on fish distribution and abundance. Indeed, Joy and Death (2001) urged caution when associating habitat characteristics with fish distribution in New Zealand, as observed distribution of diadromous species may be a function of habitat access rather than habitat preferences.

Fish community structure has the potential to influence kōura distribution and abundance in stream ecosystems through predation by single or multiple aquatic predators. Vulnerability to predation can be linked to other variables such as gape size of predators (Jones *et al.*, 2016), and habitat complexity influencing the availability of refugia such as coarse substrate and bank cover (Usio & Townsend, 2000). Vulnerability is considered highest for small crayfish (<15-25 mm OCL) as they are potentially within the gape size of many predatory fish, and high for all kōura/crayfish during moult events when their carapace is soft, and for females when they are gravid (Usio & Townsend, 2000; Jones *et al.*, 2016).

Dominant fish predators of *Paranephrops* spp. are brown trout (*Salmo trutta*) (Scott & Duncan, 1967), rainbow trout (*Oncorhynchus mykiss*) (Blair *et al.*, 2012) and eels (*Anguilla dieffenbachii* and *A. australis*) (Hicks, 1997). Olsson et al. (2006) examined the relationship between *P. planifrons* and brown trout in South Island West Coast streams and found the presence of trout influenced kōura abundance as well as size distribution, with fewer small crayfish being caught in trout streams compared to sites without trout. Usio and Townsend (2000) and Whitmore and Huryn (1999) also found that adult and young-of-the-year (YOY) *P. zealandicus* were negatively correlated with the presence of brown trout. More recently, Blair *et al.* (2012) found rainbow trout (>200 mm) consuming kōura in Lake Rotoiti, central North Island, where they comprised around 6% of their total diet. In Waikato hill-country streams, Parkyn (2000) found eels to be the main predators of *P. planifrons*, particularly on small kōura. In the same location, Hicks (1997) found kōura were the second most common food item, after mayfly larvae, in the stomachs of shortfin eels in forested streams, and in both forested and pasture streams they comprised 25-32% of longfin eel gut contents.

Jowett *et al.* (2008) reported kōura did not appear to avoid locations with potential predators such as eels, although *P. zealandicus* showed avoidance responses to chemical cues in skin mucus in eels but not trout in a laboratory experiment (Shave *et al.*, 1994). The authors hypothesised that, *P. zealandicus* responded more actively to eels given their long co-evolutionary association with this predator. Nevertheless, there is clear potential for habitat overlap between kōura and eels, increasing encounter rates and therefore predation risk for kōura populations.

Jowett and Richardson (1994) found the majority of fish in the Pohangina River, Manawatu were associated with river margins and areas less than 0.25 m deep, habitats that generally overlap with daytime kōura habitat (see Chapter 4). Eels appear to be capable of occupying a diversity of habitats (Glova, 1999), although this appears to change with size. In three small New Zealand streams studied by Glova *et al.* (1998), longfin eels <300 mm were found to occupy riffle and run areas while small shortfin eels were more prevalent in lowland rivers with deeper water. Additionally, the study found longfin eels >300 mm were associated with a variety of larger cover types, such as in-stream debris, bank overhangs and macrophyte beds, while larger shortfin eels were primarily associated with cover provided by riparian vegetation. These findings were further supported by Jellyman *et al.* (2002) who found marked changes in habitat preferences for longfin eels above and below 300 mm length, which they attributed to movement from within substrate interstices to open water. Differences in diet also occur among eel size classes. For example, in Lake Ponui, Wairarapa, Jellyman (1989) found that both small shortfin and longfin eels primarily subsisted on small invertebrates, and moved to a diet dominated by fish when they reached a size >700 mm for shortfin and >400 mm for longfin eels. In the Rakaia River in Canterbury, Sagar and Eldon (1983) established that juvenile longfin eel diet was generally similar to the proportions of prey species in the benthos, indicating they were opportunistic invertivores.

The above studies raise questions concerning the interaction strength of *Anguilla* spp. with kōura and to what degree this interaction is influenced by land-use which may impose additional stressors, such as higher water temperatures and simplified or unstable refugia that may make kōura more vulnerable to predation. Indeed, catchment vegetation, especially riparian vegetation, has been implicated as an important controlling factor for fish populations in streams through its influence on light availability, water temperature and channel stability (Hicks *et al.*, 1991). For example, comparison of predator densities between pasture and forested stream sites in the Hakarimata Ranges, Waikato, revealed that between 50-80% of pools in pastoral sites were inhabited by large or medium-sized eels compared to 10-20% in forested study sites (Hanchet, 1990). In that study, eel biomass averaged 8.6 kg in the two pastoral sites sampled compared to 1.5 kg in adjacent forested sites. The authors concluded that predation by a higher biomass of eels in pasture sites may affect prey community structure.

Similarly, Hicks and McCaughan (1997) found a seven-fold increase in eel biomass in pastoral streams compared with forested hill-country streams, concluding that differences in the fish community were likely caused by increased temperature, light, inorganic nitrogen concentrations and in-stream primary production resulting from removal of forest cover. Although *P. planifrons* were not specifically mentioned in those studies, their findings are likely relevant to predation of kōura in pasture streams where eels can reach high biomass.

Fish responses to the presence of riparian vegetation are mixed and difficult to quantify. Rowe *et al.* (2002) found riparian buffer strips generally enhanced native fish communities in steep Coromandel streams draining logged catchments, however, the abundance of eels was not affected. Holmes *et al.* (2016) assessed the influence of different types of riparian management on instream habitat condition and fish density and distribution on a Dairy Best Practice monitoring site in lowland South Canterbury. Contrary to expectations, the study found brown trout and longfin eels to be evenly distributed throughout the catchment and poorly correlated with expected fish habitat such as cover, deep water and low levels of fine sediment, all expected to be enhanced through riparian management.

Nevertheless, while riparian planting may play a role in providing additional refugia from predation for kōura (see Chapter 4), it may also influence abiotic and biotic factors affecting abundance of fish communities in hill-country pastoral streams. The impact of potential kōura predators such as eels and the overall association between fish communities on kōura distribution and abundance have not previously been examined among streams with different-aged riparian plantings. Such relationships are important to understand if the goal of restoration activities is to achieve sustainable and enduring populations of kōura at managed sites.

The aim of this chapter is to identify factors affecting the association of fish communities, particularly predatory eels, with kōura abundance in Waikato hill-country streams lined by different aged riparian planting.

Specific objectives for this chapter are:

- Determine which upstream catchment, instream and riparian factors best explain variation in eel density and size amongst sites;
- Determine the effects of riparian planting age (including no riparian planting at pastoral sites) on eel density and size compared to populations in native forest sites;
- Quantify the variation in kōura density and size explained by eel density and size amongst sites;
- Establish if the presence and density of other dominant fish groups explain variation in kōura density and size amongst sites.

Based on published work, the following hypotheses were tested in this chapter:

1) Kōura density will decrease as eel density increases, particularly eels >300mm which other studies have suggested are associated with a shift in habitat use that would increase overlap with kōura;
2) Density of *Gobiomorphus* spp., a dominant part of the benthic invertivore guild and potential predators of small kōura, will influence only YOY kōura density;
3) Distance downstream to the sea will be the dominant factor influencing eel density differences between sites with secondary influences from instream habitat or reach-scale factors linked to riparian planting age;
4) Densities of eel size classes will not be increased by riparian planting age given that other studies have reported higher eel biomass in pasture sites.

The use of the term eel or eels in the remainder of this chapter refers to both *Anguilla dieffenbachii* and *Anguilla australis* unless otherwise specified.

5.2 Methods

Sampling was undertaken within Department of Conservation Electric-Fishing Permit – 67878-RES and authorisation from Auckland/Waikato Fish and Game Council. For sites within Crown land administered by the Department of Conservation, sampling was authorised through Research and Collection Permit 70894-FAU.

5.2.1 Fish sampling

Each 50-m sample reach was divided into 5 x 10-m sub-reaches which were electric-fished in two consecutive passes using a Kainga 300 Electric-Fishing Machine (EFM) combined with substrate disturbance and brushing using a wooden-handled, soft-bristled broom. The exception was BC1 (see Figure 2.1) where the reach was 43.5 m long due to the presence of large boulder areas which were not able to be fished, resulting in sub-reaches between 5 and 10 m in length. Prior to fishing, stream conductivity was measured and best practice for voltage settings was used to optimise fishing effort while considering fish welfare (Joy *et al.*, 2013). Accordingly, pulse frequency for all sites was 60 pulse/sec with a pulse width of 2 m/sec which is commonly used to target small fish. Although effective for small fish and kōura capture, capture of eels is more suited to a reduced pulse rate (30 pulses/sec). However, given the primary target for this study was kōura, the higher pulse rate was chosen to maximise the potential to capture kōura. Anode size (small or large) varied between sites and was a function of stream type and habitat.

All sites were fished on the same day with the exception of O1 which was fished over two consecutive days. Sampling was primarily conducted by a team of four comprising one fisher, one person on a stop-net and two team members sweeping to dislodge fish and create flow into the hand-held stop net downstream. In smaller streams, or where habitat was less complex and space was constrained, fishing was occasionally conducted using three people with only one team member sweeping while the fisher also utilised a broom to create flow behind the EFM anode. A net (3 mm x 3 mm mesh) was installed across the downstream end of each sub-reach prior to being fished to capture any fish that avoided hand nets.

Areas of 1-1.5 m^2 were fished in bands moving progressively upstream across the width of the channel. Sweeping occurred during shocking and also for approximately 20 sec after shocking had ceased. In areas with predominantly cobble or gravel substrate, sweeping was supplemented with substrate disturbance by feet or hands after shocking was complete. The stop-net was checked after each area had been fished and all fish were placed in corresponding buckets on the stream edge. The downstream fixed-net was checked after two consecutive passes had been completed and any fish were combined into the sub-reach sample. No bias was placed on any particular habitat during sampling with equal effort applied across all sample reaches.

All captured eels were measured for total length, however, only those c. >200 mm were identified to species level, as they represented potential predators of large kōura. All eels <200 mm were classified as elvers. All captured eels were anesthetised using diluted AQUI-S, for measurement and then returned to a recovery bucket or eel bag in the stream away from the electric field. All other fish were identified and counted but not measured. Following measurement of eels, all fish were returned to the source sub-reach after the downstream net had been moved to the next upstream sub-reach to prevent movement and double-counting.

5.2.2 Statistical analyses

All fish species combined for each site were used to calculate total fish density but only fish that appeared in at least 50% of study sites were used in more detailed analyses (i.e., *Gobiomorphus* spp. and eels). Three bully species (red fin bully - *Gobiomorphus huttoni*, Crans bully - *Gobiomorphus basalis* and common bully - *Gobiomorphus cotidanus*) were difficult to distinguish for small fish, but have similar habitat and feeding behaviour and preferences (McDowall, 2000) so were amalgamated for analysis. Similarly, as elver species were not differentiated and longfin eels heavily dominated the catch of those large enough to be identified (see Section 5.3), both eel species were combined in these analyses.

Jellyman *et al.* (2002) found longfin eels moved from strong associations with substrate to open water habitats at ~300 mm in length, while this movement occurred for shortfin eels ~200 mm. Accordingly, and given the dominance of longfin eels, total eel density (no. per m^2) was further categorised into ≤300 mm and >300 mm size classes for additional analyses, as the larger size class was considered to have greater potential to influence kōura density. Prior to analyses, the relationship between mean eel length (mm) and median eel length (mm) was tested and found to be highly correlated ($r_s = 0.917$), so mean eel length (mm) was selected to test for size differences among riparian planting age groups.

Spearman rank correlations were used to examine the strength and direction of association between densities (no. per m^2) of total and YOY kōura with eels (total, ≤300 mm and >300 mm), total bullies and all fish combined. Additionally, Spearman rank correlations were used to examine the strength and direction of association between eel densities and distance downstream to the sea (km).

The step-wise distance-based linear model procedure (DistLM, PRIMER 7.0.13) used previously to determine which predictor variables best explained the dissimilarities in total kōura density across all sites (see Table 4.14), was also used to establish if the addition of the six fish categories (total fish, bullies, and eels total, ≤300 mm, >300 mm densities and average eel length) increased the variation explained in the selected overall kōura model. The response variable in this model was the between site dissimilarity matrix based on the Bray-Curtis distances of transformed (log x+1) total kōura density data (see Chapter 4).

DistLM was also used to test which upstream catchment, riparian and instream habitat predictor variables best explained the dissimilarities in total eel density between the 26 sites (native, riparian and pasture) and produced the most parsimonious model. The predictor variables used were the same as those described in Chapters 3 and 4, with the addition of downstream distance to the sea. The response variable in these models was the between-site dissimilarity matrix based on the Bray-Curtis distances between transformed (log x+1) total eel density data. Variables identified as highly inter-correlated based on draftsman plots (see Chapter 3 and 4) were excluded from this analysis.

Using the total eel density resemblance matrix, marginal tests and best-fit selection processes were performed individually on transformed, normalised upstream catchment, riparian and instream habitat predictor variables. Adjusted R^2 was used as the criterion for variable selection as it takes into account the number of parameters and excludes predictor variables if they add no more to the explained sum of squares. In Chapters 3 and 4, variables were only included in the manual and forward models if they explained >3% of the variation in the model however, the best-fit variable selection process produces only total adjusted R^2 and cumulative R^2 values representing all the variables contained in the model rather than for individual variables, making this threshold difficult to ascertain. To minimise differences, best-fit model selection was based on those models that contained approximately half the variables included in the original analysis and contained variables which were logical based on knowledge of streams, the interaction of habitat variables, and eel and kōura ecology. Learnings from both Chapter 3 and Chapter 4 were also used as a guide for model selection.

The variables contained in each of the separate selected models (upstream catchment, riparian or instream habitat) were then combined and reanalysed to establish the overall best-fit model option to explain dissimilarities in total eel densities across the study sites. Spearman rank correlations were used to examine the strengths and directions of association between total eel density (no. per m^2) and the variables contained in the overall best-fit model. The overall selected best-fit model represented the most logical and parsimonious model accounting for most variation in total eel density (no. per m^2) between sites.

Effects of riparian planting age on eels

Two-way PERMANOVA (PRIMER 7.0.13) was used to test for overall between-site differences in total eel density using riparian age groups (excluding native sites) and site spatial cluster as factors. Analysis used the dissimilarity matrix based on the Bray-Curtis distances between transformed total eel density (no. per m^2) data.

Non-parametric Kruskal-Wallis test (STATISTICA v13.3) was used to determine ranked differences in fish population characteristics among different-age classes of riparian plantings. Riparian sites were classified into age blocks of 1 = pasture (i.e. zero riparian age), 2 = 1-5 years or recent riparian age, 3 = 6-10 years or mid riparian age, 4 = 11-15 years or mid-late riparian age, and 5 = 16+ years or old riparian. Native sites were excluded from this analysis to test the null hypothesis that there would be no difference in eel population characteristics between riparian-planting age groups at sites formerly or currently grazed.

Non-parametric Mann-Whitney U tests (STATISTICA v13.3) were performed on two combinations of riparian age classes for fish population variables to establish whether sites with older riparian plantings exhibited fish population characteristics similar to native forest sites. The paired age classes compared were 11-15 years (mid-late) versus native forest sites and 16+ years (old) versus native forest sites.

5.3 Results

5.3.1 Fish community characteristics

A summary of fish community characteristics across different treatment groups types is shown in Table 5.1. For fish data for individual sites refer to Appendix 10.

In total 1,605 fish were caught across the 26 study sites at an average density of 0.69 m^2. Bullies comprised 49% of the total catch followed by eels (36%). Mean downstream distance to the sea for native, riparian and pasture sites was similar (95-98 km; Table 5.1).

A total of 788 bullies were captured at 17 study sites representing a combination of red fin, common and Crans bullies. Of these, 89 or 11% were unidentified to species level due to their small size. Larger Crans and common bullies could not be differentiated due to taxonomic uncertainties. Average bully density across all sites where captured (n = 17) was 0.49 per m^2. The next most abundant group was eels, found at all sites with 572 captured at an average density of 0.26 per m^2. Of those, 331 were classified as elvers measuring between 40 mm to 200 mm total length, and were not identified to species level. Of the larger eels that were identified, 236 were longfin eels and five were shortfin eels. A total of 477 eels were ≤300 mm in length and 95 were >300 mm in length. Average eel length across all sites was 241 mm and median length was 202 mm. The smallest eel captured was 40 mm and the largest 1010 mm total length.

A total of 31 trout (*Salmo trutta* and *Oncorhynchus mykiss*) were captured but these species only occurred at three sites (R1, KA1 and MK1; see Figure 2.1); they had an average fork length of 56.9 mm. One large *O. mykiss* was captured at site R1 (265 mm fork length).

Altogether, 211 Galaxiidae spp. were captured at 12 of the study sites with the majority (125 fish) at site OM1 (see Figure 2.1). Their length varied but many were YOY between 40 mm and 60 mm total length. Some larger specimens (60–130 mm) were also captured occasionally, comprising inanga (*Galaxias maculatus*), banded kōkopu (*Galaxias fasciatus*) and one giant kōkopu (*Galaxias argenteus*) measuring 210 mm at WK1 (see Figure 2.1). One lamprey (*Geotria australis*) ammocoete was captured at site R1, along with one torrentfish (*Cheimarrichthys fosteri*) at site MK1 and one smelt (*Retropinna retropinna*) at site TU2.

Table 5.1: Mean, standard deviation (SD) and minimum and maximum values for site distances to sea and for fish characteristics captured at study sites. Pasture n = 3, riparian n = 20, native n = 3.

Variable	**Pasture**				**Riparian**				**Native**			
	Mean	SD	Min	Max	Mean	SD	Min	Max	Mean	SD	Min	Max
Distance to sea (km)	97.50	± 75.30	13.00	157.40	95.94	± 71.33	1.14	183.00	95.00	± 76.60	8.60	154.50
Total eels (no. per m^2)	0.27	± 0.38	0.04	0.71	0.28	± 0.31	0.03	1.23	0.06	± 0.07	0.02	0.14
Eels >300mm (no. per m^2)	0.06	± 0.07	0.01	0.14	0.04	± 0.05	0.00	0.18	0.01	± 0.02	0.00	0.40
Eels ≤300 mm (no. per m^2)	0.21	± 0.31	0.01	0.57	0.24	± 0.30	0.00	1.22	0.04	± 0.05	0.01	0.10
Eel average length (mm)	291.00	± 151.30	195.30	465.10	242.96	± 142.00	87.66	549	178.90	± 60.50	110.00	223.30
Total bullies (no. per m^2)	1.00	± 0.59	0.53	1.17	0.29	± 0.66	0.03	2.94	0.07	± 0.09	0.04	0.18
Total fish density (no. m^2)	0.85	± 0.33	0.60	1.23	0.75	± 1.11	0.03	3.97	0.14	± 0.17	0.02	0.33

5.3.2 Fish associations with kōura density

The densities of eels ≤300 mm and >300 mm and total bullies, were not significantly correlated with total kōura density. However, the densities of total eels (r_s = -0.40, p = 0.04) and total fish (r_s = -0.58, p = 0.001) were significantly and inversely correlated to total kōura density.

The density of eels >300 mm was not significantly correlated with total fish density, however, the density of eels ≤300 mm (r_s = 0.70, p = 0.001) and total bullies (r_s = 0.71, p = 0.001) were, indicating these two groups were primarily responsible for differences in total fish density between sites.

For YOY kōura density, no significant correlations were detected with total eels, eels ≤300 mm, eels >300 mm, or total bully densities, however, total fish density was significantly negatively correlated with YOY kōura density (r_s = -0.60, p = 0.002).

The relationship between kōura and total eel density is presented in Figure 5.1. Although the correlation was statistically significant, the linear regression explained only 13% of the variation in kōura density. Nevertheless, Figure 5.1 indicates kōura densities were consistently <0.5 m^2 where eel densities were >0.3 m^2.

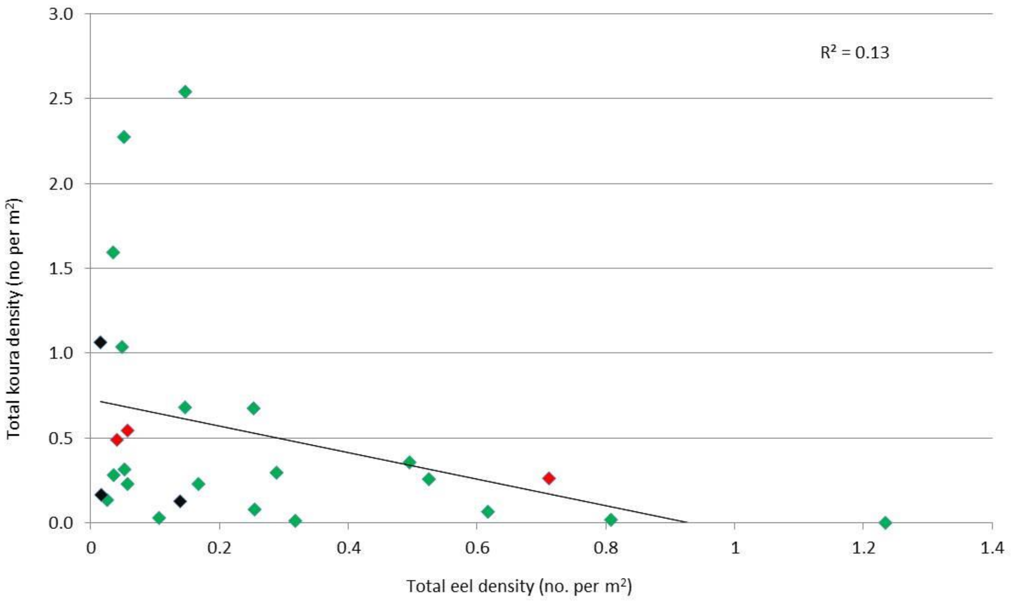

Figure 5.1: Kōura density as a function of total eel density for 20 riparian sites (green), 3 pasture sites (red) and 3 native forest sites (black) in Pirongia, Whaingaroa and Whatawhata. Linear trend line is in black with associated R^2 value.

The relationship between total kōura and total fish density is presented in Figure 5.2. The relationship was best expressed as a logarithmic function which accounted for 26% of the variation in total kōura density, with the highest number of kōura found at sites with the lowest fish density. Fish densities were highest at sites OM1 (3.97 per m^2) which had no kōura present, and at TU1 (3.58 m^2) which had 0.06 kōura per m^2, and very high densities of both bullies and small eels.

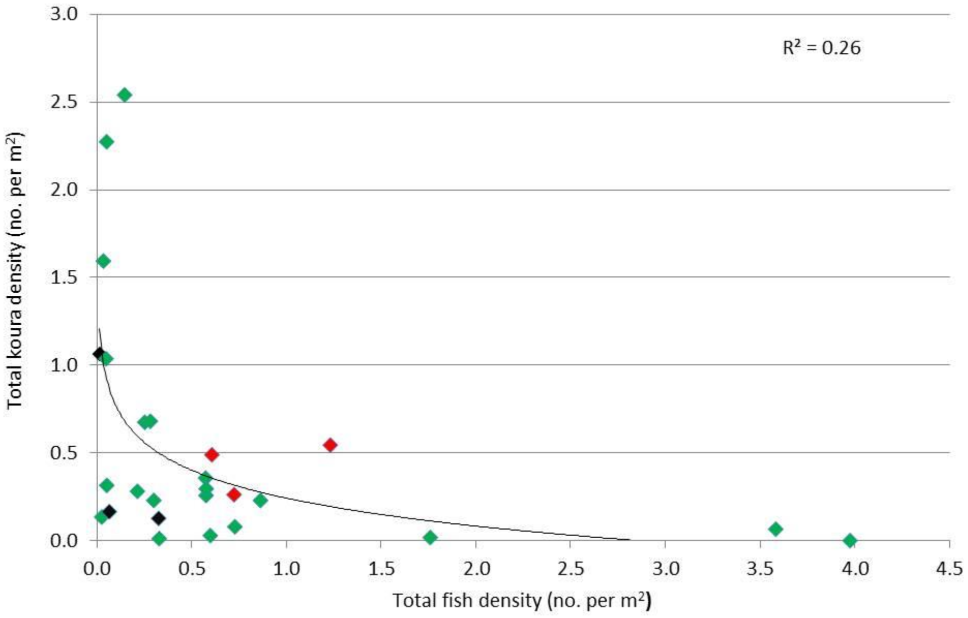

Figure 5.2: Kōura density (no. per m^2) as a function of total fish density (no. per m^2) for 20 riparian sites (green), 3 pasture sites (red) and 3 native forest sites (black) in Pirongia, Whaingaroa and Whatawhata. Logarithmic trend line is in black with associated R^2 value.

Logarithmic regression indicated total fish density explained 30% of the variation in the relationship with YOY kōura (Figure 5.3.). Although YOY kōura were not found at three sites in this study, these have been included in this analysis as fish density may have contributed to their absence. Fish densities at these sites without YOY kōura varied from <0.5 to 3.97 per m^2. Generally, YOY densities were higher where total fish density was lower. In contrast, YOY kōura density was mostly <0.5 per m^2 where fish density was >0.5 per m^2.

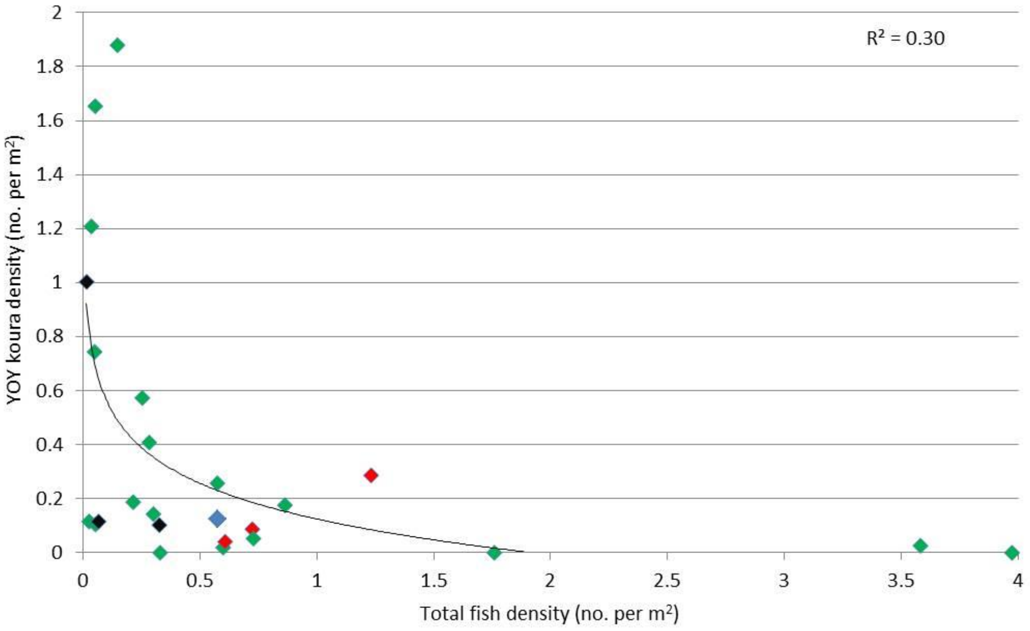

Figure 5.3: YOY kōura density (no. per m^2) as a function of total fish density (no. per m^2) for 20 riparian sites (green), 3 pasture sites (red) and 3 native forest sites (black) in Pirongia, Whaingaroa and Whatawhata. Logarithmic trend line is in black with associated R^2 value.

The DistLM model selected to best explain variation in kōura density dissimilarities between sites (Table 4.14) was used to examine whether, individually, the addition of each of six fish variables would improve its explanatory power. Marginal test values for each of the six fish categories are reported in Table 5.2 and tests for upstream catchment, instream habitat and riparian variables for the kōura density dissimilarity matrix are reported in Appendix 7. All variables used accounted for ≥3% variation but total fish, eels ≤300 mm and total eels had the highest R^2 (14-19% of variation).

Table 5.2: Marginal tests for fish categories using the kōura density (no. per m^2) between site dissimilarity matrix as the response variable.

Variables	**Marginal test (Cumulative R^2)**
Total eels (no. per m^2)	0.14
Eels ≤300 mm (no. per m^2)	0.15
Eels >300 mm (no. per m^2)	0.04
Mean eel length (mm)	0.03
Total bullies (no. per m^2)	0.09
Total fish density (no. per m^2)	0.19

The addition of each fish category to the overall kōura density model only improved the total variation accounted for by between one and two percent (Table 4.14). Total eels and eels ≤300 mm densities featured in their respective best-fit models with both replacing average stream bank slope. Total fish density featured in its best-fit model and replaced “other” vegetation. Density of eels >300 mm, and total bullies and average eel length did not feature in their respective best-fit models.

Table 5.3: DistLM analysis summary of the cumulative effect of instream habitat, physicochemical, riparian, upstream catchment and fish density variables on kōura density (no. per m^2) dissimilarity among 26 study sites using best-fit selection processes and adjusted R^2 as the selection tool. Increase in cumulative R^2 indicates change due to inclusion of fish variable to final model kōura model; X = presence of variable in model; + = addition of fish variable to final kōura model from Chapter 4; Grey fill = variable not selected for generation of model.

Variable	**Kōura**	**+Total eels**	**+Eels ≤300 mm**	**+Eels >300 mm**	**+Eel average length (mm)**	**+Total bullies**	**+Total fish density**
Macrophyte cover (%)	X	X	X	X	X	X	X
Root complexes (m^2)	X	X	X	X	X	X	X
Cobble substrate (%)	X	X	X	X	X	X	X
Sand substrate (%)	X	X	X	X	X	X	X
Number of saplings (per m^2)	X	X	X	X	X	X	X
Dissolved oxygen (mg/L)	X	X	X	X	X	X	X
Catchment area (ha)	X	X	X	X	X	X	X
“Other” vegetation (%)	X	X	X	X	X	X	-
Ashes older than Taupō eruption (%)	X	X	X	X	X	X	X
Average stream bank slope (°)	X	-	-	X	X	X	X
Total eels (no. per m^2)		X					
Eels ≤300 mm (no. per m^2)			X				
Eels >300mm (no. per m^2)				-			
Eel average length (mm)					-		
Total bullies (no. per m^2)						-	
Total fish density (no. per m^2)							X
Total number of variables	*10*	*10*	*10*	*10*	*10*	*10*	*10*
Cumulative R^2	*0.83*	*0.84*	*0.85*	*0.84*	*0.84*	*0.84*	*0.85*
Increase in cumulative R^2	-	*0.01*	*0.02*	*0.01*	*0.01*	*0.01*	*0.02*

5.3.3 Factors affecting eel density

Distance downstream to the sea was not significantly correlated with eel density >300 mm but it was strongly and negatively correlated with densities of total eels (r_s = -0.67, p = <0.001) and those ≤300 mm length (r_s = -0.66, p = <0.001).

DistLM was used to examine which upstream catchment, instream habitat and riparian factors best explained variation in total eel density expressed as a resemblance matrix between sites. Results from marginal tests are reported in Appendix 11.

The best-fit upstream catchment model for total eel density contained six variables that accounted for 60% of the variation between study sites. The model included three types of catchment vegetation cover (percentage grass, percentage “other” vegetation and percentage of upstream riparian buffer in woody vegetation), two types of geology (percentage mudstone and sandstone), and distance downstream to the sea.

The best-fit riparian model for total eel density contained six variables that explained 38% of the variation between study sites. Four variables were associated with riparian vegetation cover (number of saplings, basal area and percentage leaf litter and bare ground), and two were associated with the more structural components of streams, namely average stream bank slope and buffer width.

The best-fit instream habitat model for total eel density contained ten variables that explained 62% of the variation between study sites. Five variables were associated with hydraulic habitat (thalweg depth, reach gradient, and run/riffle/pool habitat), three with inorganic substrate size (cobble, gravel, silt) and two with organic features, namely macrophyte cover and root complexes.

A total of 22 variables were combined from the upstream catchment, riparian and instream habitat models described above, and reanalysed using the best-fit selection process. Correlations for the selected variables in the overall best-fit model with total eel density to identify the direction of effect revealed six of the variables had statistically significant individual correlations with eel density

(Table 5.4). The overall best-fit model explained 85% of variation in total eel density between sites and contained a total of 11 variables: three upstream catchment variables (percentage grass vegetation (+), percentage upstream buffer in woody vegetation (-) and distance to sea (-)); two riparian variables (percentage bare ground (+) and number of saplings (+)); and six instream variables related to substrate (percentage gravel (weak -) or silt (+)); hydraulic habitat (percentage run (+) or riffle (-)) and organic cover (cover by macrophytes (+) or root complexes (-)).

Table 5.4: Spearman rank correlations and associated p values for each variable in the overall final model with total eel density. Significant p values for correlations (<0.05) are highlighted in bold.

Variable contained in overall total eel density model	**r_s value**	**p value**
Grass vegetation (%)	+0.40	**0.043**
Riparian buffer in woody vegetation (%)	-0.25	0.217
Number of saplings (per m^2)	+0.13	0.521
Bare ground (%)	+0.23	0.140
Root complexes (m^2)	-0.40	**0.043**
Gravel substrate (%)	-0.03	0.872
Silt substrate (%)	+0.24	0.226
Riffle habitat (%)	-0.42	**0.030**
Run habitat (%)	+0.39	**0.049**
Macrophyte cover (%)	+0.57	**0.002**
Distance to sea (km)	-0.67	**0.001**

The overall model for total eel density was compared to the overall kōura density model to ascertain if there were commonalities, and to explore the possibility that a variable from the kōura model may be acting as a surrogate for eel density (Table 5.5). The two models had only three variables in common, namely root complexes, number of saplings and percentage macrophyte cover. Given these differences, and the contrasting effect directions of some (e.g., root complexes and macrophyte cover), it is unlikely that variables explaining dissimilarities in kōura density were acting as surrogates for eel density, and therefore this issue was not explored further.

Table 5.5: Comparison of the DistLM analysis summaries of the cumulative effect of instream habitat, upstream catchment and riparian variables on total eel density (no. per m^2) and the cumulative effect of instream habitat, upstream catchment, physicochemical and riparian variables on total kōura density (no. per m^2) dissimilarity among 26 study sites using best-fit selection processes and adjusted R^2 as the selection tool. X = presence of variable in model.

Variable	**Eel model**	**Kōura model**
Grass vegetation (%)	X	-
Riparian buffer in woody vegetation (%)	X	-
Number of saplings (per m^2)	X	X
Bare ground (%)	X	-
Root complexes (m^2)	X	X
Gravel substrate (%)	X	-
Silt substrate (%)	X	-
Riffle habitat (%)	X	-
Run habitat (%)	X	-
Macrophyte cover (%)	X	X
Distance to sea (km)	X	-
Cobble substrate (%)	-	X
Sand substrate (%)	-	X
Dissolved oxygen (mg/L)	-	X
Catchment area (ha)	-	X
"Other" vegetation (%)	-	X
Ashes older than Taupō eruption (%)	-	X
Average stream bank slope (°)	-	X
Total number of variables	*11*	*10*
Cumulative R^2	*0.85*	*0.83*

5.3.4 Effects of riparian planting age on fish

PERMANOVA revealed no significant effect of riparian age group (Pseudo-$F_{4, 22}$ = 1.204, p = 0.317) or site spatial cluster (Pseudo-$F_{2, 22}$ = 1.952, p = 0.124) on total fish density (no. per m^2) dissimilarity among sites. There was no significant interaction between site spatial cluster and riparian age group (Pseudo-$F_{4, 22}$ = 0.482, p = 0.890).

For total eel density (no. per m^2), dissimilarity among sites, PERMANOVA revealed no significant effect of riparian age group (Pseudo-$F_{4, 22}$ = 0.948, p = 0.484), however, site spatial cluster was significant (Pseudo-$F_{2, 22}$ = 5.702, p = 0.005). Pairwise tests for site spatial clusters revealed significant differences between Whaingaroa and Pirongia sites (t = 3.034, p = 0.005), and no significant difference between Whaingaroa and Whatawhata (t = 1.900, p = 0.104) or Whatawhata and Pirongia (t = 1.339, p = 0.189). There was no significant interaction between site spatial cluster and riparian age group (Pseudo-$F_{4, 22}$ = 0.533, p = 0.781).

Kruskal-Wallis test to determine the effect of different riparian planting-age groups on total fish density, eel population characteristics and total bully density revealed no significant differences (Table 5.6). For completeness, box plots for these variables are presented in Figures 5.4 to 5.9.

Table 5.6: Results of Kruskal-Wallis analysis for differences in total fish, eel or bully population characteristics between riparian planting age groups: 0 years = pasture, 1-5 years old (recent), 6-10 years old (mid), 11-15 years old (mid-late) and 16+ years old (old). No p values were statistically significant at <0.05.

Variable	**H Statistic**	**p value**
Total eels (no. per m^2)	3.93	0.415
Eels ≤300 mm (no. per m^2)	3.84	0.428
Eels >300mm (no. per m^2)	0.48	0.975
Average eel length (mm)	5.72	0.221
Minimum eel length (mm)	6.65	0.155
Maximum eel length (mm)	3.20	0.524
Median eel length (mm)	8.70	0.689
Total bullies (no. per m^2)	4.06	0.398
Total fish density (no. per m^2)	6.72	0.151

Total fish density showed little variation in the pasture sites, or in the mid-aged and old plantings, but it was highly variable in the mid-late and recent planting groups. With the exception of mid-late plantings, all means were relatively similar and fell between 0.16 and 0.85 fish per m^2.

Also, with the exception of mid-late plantings, total and ≤300 mm eel density showed little pattern related to riparian age group. Recent and old plantings showed similar mean densities and low variability for eels >300 mm, while pasture, mid and mid-late aged plantings had similar mean densities but high variability. Old riparian planting sites had the lowest mean for average eel length and low variability. All other planting categories had similar mean lengths and high variability. With the exception of mid-aged riparian planting sites, very few differences were evident between planting age groups for total bully density, although variability was higher in recent plantings. The mid-aged category result reflects the lack of bullies captured at any of the sites in this group.

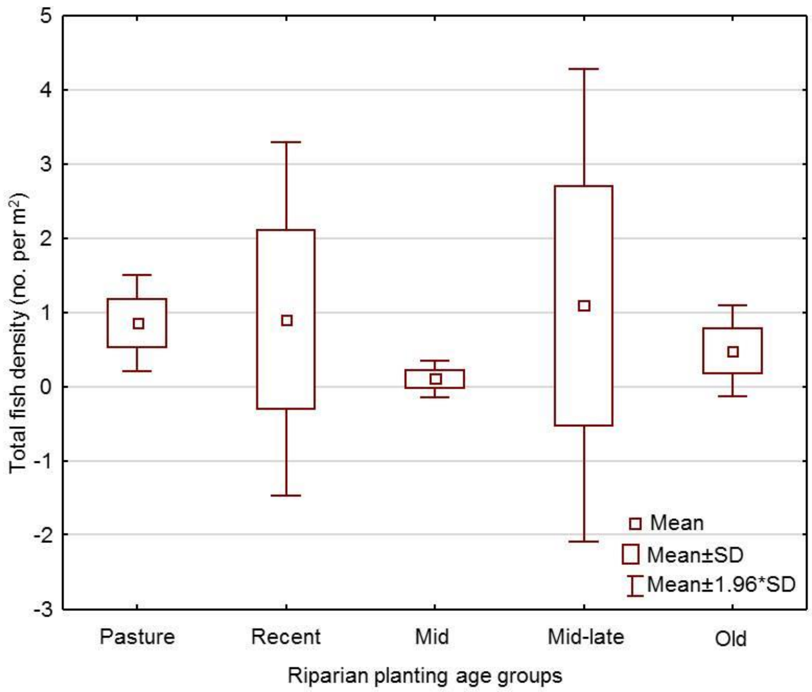

Figure 5.4: Box plot of total fish density (no. per m^2) by riparian planting age groups for 23 Pirongia, Whaingaroa and Whatawhata stream sites.

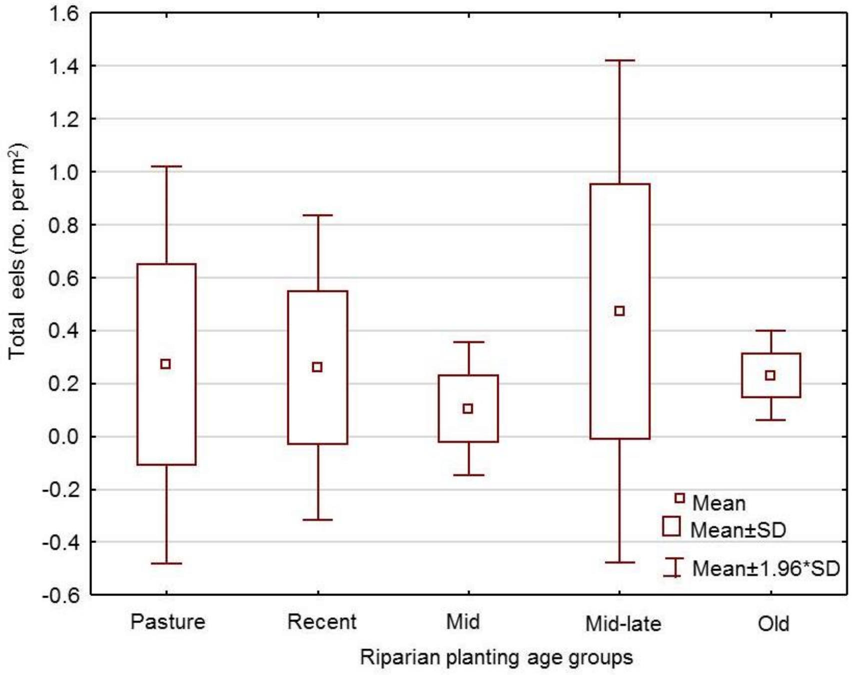

Figure 5.5: Box plot of total eel density (no. per m^2) by riparian planting age groups for 23 Pirongia, Whaingaroa and Whatawhata stream sites.

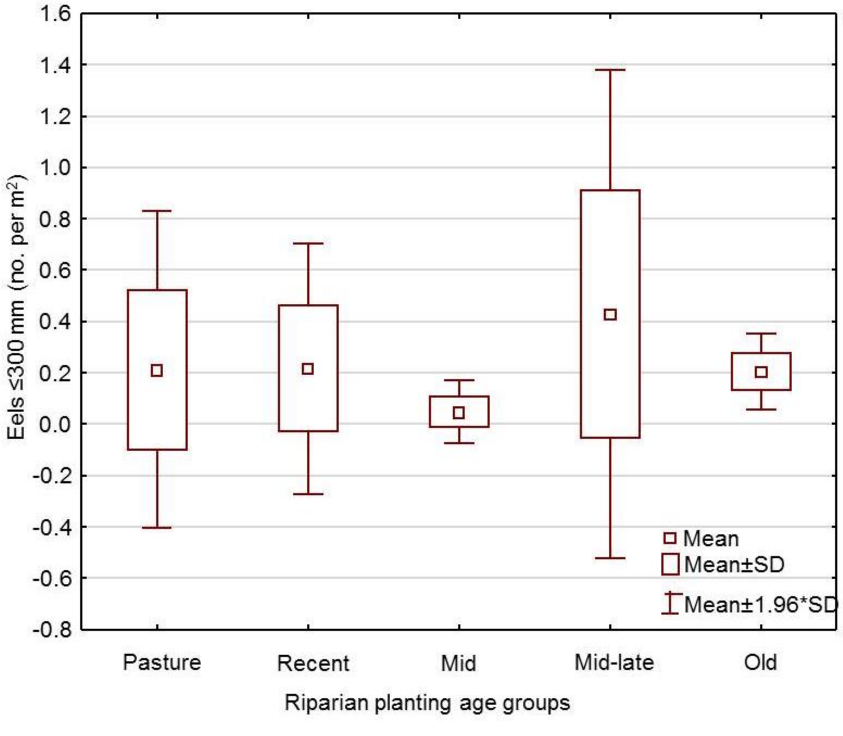

Figure 5.6: Box plot of eel density ≤300 mm (no. per m^2) by riparian planting age groups for 23 Pirongia, Whaingaroa and Whatawhata stream sites.

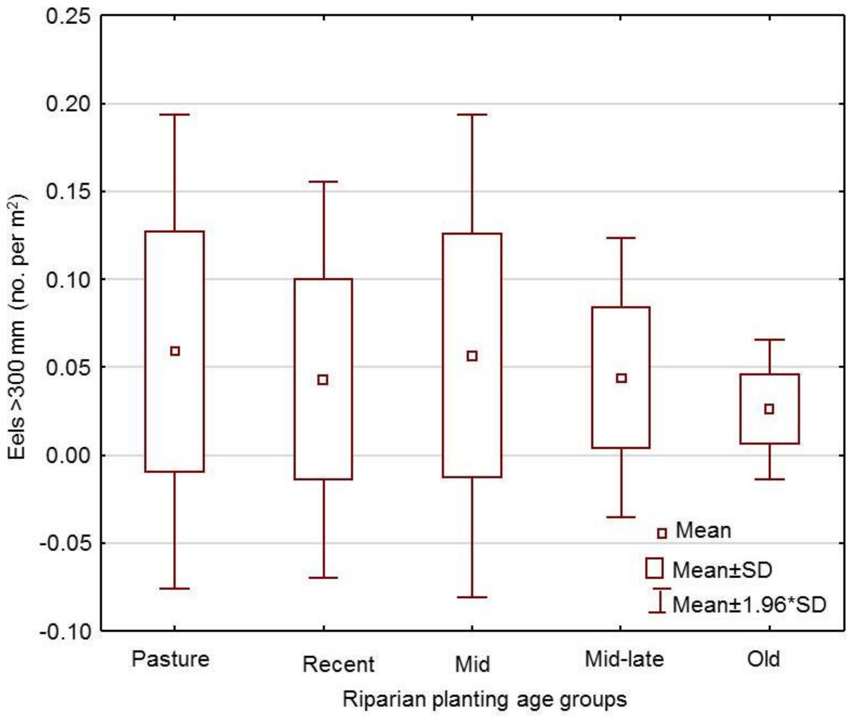

Figure 5.7: Box plot of eel density >300mm (no. per m^2) by riparian planting age groups for 23 Pirongia, Whaingaroa and Whatawhata stream sites.

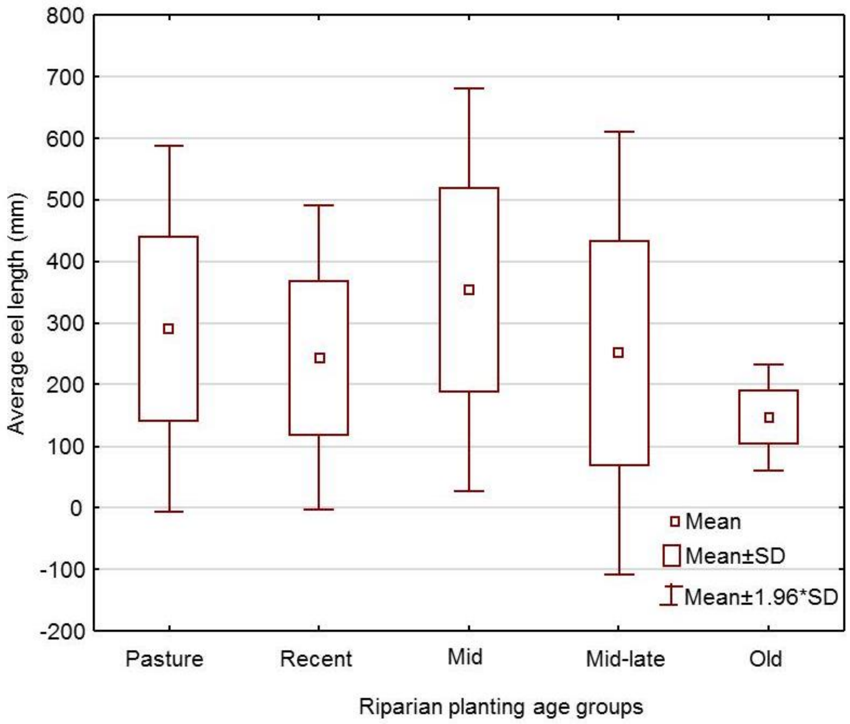

Figure 5.8: Box plot of average eel length (mm) by riparian planting age groups for 23 Pirongia, Whaingaroa and Whatawhata stream sites.

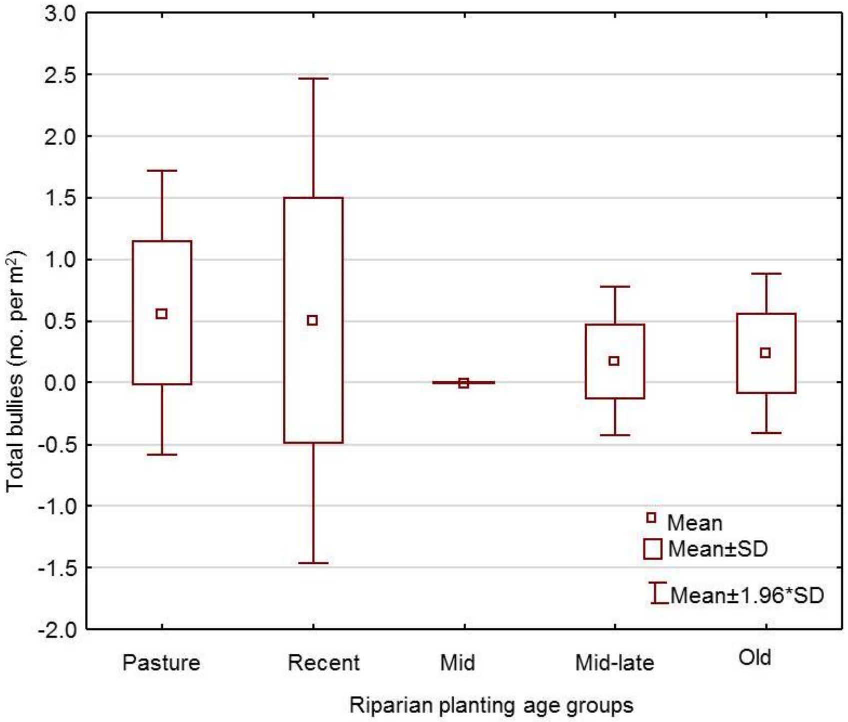

Figure 5.9: Box plot of total bully density (no. per m^2) by riparian planting age groups for 23 Pirongia, Whaingaroa and Whatawhata stream sites.

Mann-Whitney U tests for differences between mid-late and old riparian age groups with native forest sites also revealed no significant differences for total fish density, eel population characteristics or total bully density (Table 5.7), however, total and ≤300 mm eel densities did show some differences between old and native sites although these were not statistically significant.

Table 5.7: Results of Mann-Whitney U tests for total fish density, eel population characteristics and total bully density between 11-15 years old planting sites (mid-late) versus native forest sites and 16+ years old planting sites (old) versus native forest sites. Mid-late riparian sites n = 5, old riparian sites n = 4, native forest sites n = 3. No p values were statistically significant at <0.05.

Variable	**11-15 years (Mid-late) vs Native**		**16+ years (Old) vs Native**	
	U statistic	**p value**	**U statistic**	**p value**
Total eels (no. per m^2)	1.00	0.074	0.00	0.051
Eels ≤300 mm (no. per m^2)	2.00	0.136	0.00	0.051
Eels >300mm (no. per m^2)	1.00	0.074	4.50	0.723
Average eel length (mm)	7.00	1.000	3.00	0.376
Minimum eel length (mm)	7.00	1.000	4.50	0.723
Maximum eel length (mm)	4.00	0.371	4.00	0.595
Median eel length (mm)	5.50	0.655	5.50	1.000
Total bullies (no. per m^2)	7.00	1.000	5.00	0.859
Total fish density (no. per m^2)	5.00	0.571	2.00	0.215

5.4 Discussion

The focus of this chapter has been on: i) determining associations between kōura density and size and fish communities in small streams of the western Waikato; ii) determining the relative influence of upstream catchment and/or reach-scale instream habitat and riparian factors on density and size of eels, a known predator of kōura; and iii) determining the effects of different riparian planting age on fish communities compared to pastoral and native forest streams.

Sites were clustered across three spatial areas, each with native forest and pasture sites represented. Although spatial bias between site clusters was not evident in analyses of instream habitat (Chapter 3) and kōura densities (Chapter 4), significant differences were detected for analysis of eel densities in this chapter, most likely reflecting close proximity to sea of the Whaingaroa sites compared to other sites.

With the exception of two sites (TP3 at 38.7 m^2 and TP5 at 38.1 m^2), all sites had sample areas >50 m^2, the minimum area suggested in Jowett and Richardson (1996) to establish representative density estimates. With the exception of OM1

(3.97 m^2) and TU1 (3.58 m^2), all sites fell within the average fish density range (0.005-2.0 fish m^2) also established in Jowett and Richardson (1996) for 38 medium to large New Zealand rivers. However, the 50-m reach length used in the current study is smaller than the 150 m recommended for single-pass electric-fishing by David *et al.* (2010) for representing fish community richness, with >90% of species recorded within 150 m. In this study, given the primary target was kōura, effort was put into fishing each reach twice to provide more robust estimates of fish density, rather than fishing longer sample reaches once to detect additional fish species.

While the current study sought to quantify the density of dominant fish species and size of eels in terms of their effects on kōura, results need to be applied with caution. Fish are more mobile than kōura, so the assessment of the fish community and its potential influence on kōura at a 50-m reach scale is limited. Additionally, the chosen reach may not have contained suitable habitat for some fish species during the day-time, however, this habitat may have existed in close proximity enabling fish to still influence the sample reach during upstream and downstream movements. Finally, the surveys represent a snapshot in time and, given the nocturnal habits and seasonality of diadromous fish migrations, community composition derived from daytime electric-fishing may change during the course of a day and seasonally. Regardless of these limitations, the data provides valuable insights into the likely associations between fish and kōura densities during the day in summer in contrasting Waikato hill-country streams that are likely typical of similar streams throughout the North Island.

5.4.1 Fish populations in study streams

Fish communities across the 26 study sites were dominated numerically by eel and bully species, although Galaxiidae spp. were prevalent in the Whaingaroa sites which were closer to the sea. Jowett and Richardson (1995), Joy and Death (2001) and Hanchet (1990) all reported similar findings with catches from small streams in Taranaki or Waikato dominated by eel and bully species, particularly for streams located further inland.

Trout were found at three of the study sites and, with the exception of one fish (265 mm), all were <86 mm (fork length). Electric-fishing generally underestimates numbers of trout >100 mm fork length (Habera *et al.*, 2010; Meyer & High, 2011), indicating that the absence of large trout may have partly been a function of sampling method. Although a potential kōura predator, trout occurred at so few sites that they were not a major component of fish communities in this study and detailed analysis was not undertaken. Various studies, such as Olsson *et al.* (2006), Usio and Townsend (2000) and Blair *et al.* (2012), have documented negative relationships between trout and kōura density, however, the impact of trout populations on kōura in these Waikato hill-country streams seems likely to be minor given their sparse distribution.

With the exception of five eels, all eels >200 mm were longfin suggesting they comprised the majority of the total catch. Shortfin abundance is generally higher in the lower reaches of streams close to the coast (Glova, 1999), so this species may therefore have comprised a higher proportion of the unidentified elvers (eels ≤200 mm) in Whaingaroa sites. The most important factors influencing distribution of juvenile longfin eels in lowland streams was velocity and substrate (Glova *et al.*, 1998) suggesting longfin elvers were likely more common on the steeper, elevated sites further inland associated with Pirongia and Whatawhata.

5.4.2 Association of fish with kōura

Parkyn (2000) identified eels as the primary predator of kōura in Waikato hill-country streams. However, she noted, at least for pasture streams, that both kōura and eels were positively associated with instream cover, with kōura abundant in reaches where eel densities where also high, although the relationship between eel density and kōura was not specifically examined. In the current study, total fish density was significantly and negatively correlated with total and YOY kōura density accounting for 25-30% of the variation in YOY kōura densities. Total fish density was significantly positively correlated to both eels ≤300 mm and total bully densities, indicating that these two groups of fish are potentially influencing these relationships with kōura rather than eels >300 mm (no significant correlation).

Total eel density was also negatively correlated to total but not YOY kōura density, and when added to the overall kōura density model total eel density explained very little additional variance. The model selection of eels ≤300 mm, and their positive correlation with total fish density, suggests that the abundance of small eels, have a negative influence, at least on smaller kōura, although this could also reflect lower kōura numbers coupled with higher small eel numbers in sites at low elevation and/or closer to the sea.

The lack of correlation of large eel densities with total kōura densities, coupled with the absence of eel >300 mm density in the overall kōura model, suggests larger eels have a minimal influence on either total or YOY kōura in the current study. Indeed, large eels along with kōura were both prevalent at many sites. The relative abundance of kōura appeared to be more closely linked to availability of suitable habitat rather than the presence of eels. Field observations also confirmed the ability of both small and large eels to utilise a diverse range of habitats at the reach-scale within a site, with daytime habitat selection appearing to be driven by habitat availability, and at times apparently coinciding with preferred kōura habitat.

Given the wide range of habitats used, reach-scale overlap between kōura and eels seems inevitable at some stage. The importance of habitat complexity in providing refugia for kōura is well recognised, including refugia from predation (Usio & Townsend, 2000). In the current study, root complexes were positively associated with kōura and negatively associated with eels suggesting they may provide refugia from predation at study sites. Indeed, where sufficient refugia are available, suitable conditions for co-existence of kōura with eels likely prevail, at least at densities which reflect natural eel recruitment. Density thresholds identified in this study indicate that kōura densities were consistently <0.5 m^2 where total eel densities were >0.3 m^2 and were also <0.5 m^2 where total fish densities were >1.25 m^2. For YOY kōura, densities were consistently <0.3 m^2 where total fish density was >0.75 m^2.

Only three factors identified as influencing kōura density overlapped with those predicting eel density in the models tested. Number of saplings was weakly positively associated with kōura and weakly negatively associated with eels. This contrasting response possibly indicates preference of kōura for reaches with riparian cover, as identified previously in Parkyn and Collier (2004) and Jowett *et al.* (2008), and the ability of eels, particularly longfins, to utilise a wide variety of habitats, not necessarily associated with riparian cover (Glova *et al.*, 1998). Root complexes were positively associated with both total and YOY kōura and significantly negatively associated with eels, supporting previous findings in Chapter 4 of the observed importance of root complexes as kōura habitat and highlighting their potential role as refugia from eel predation. Conversely, macrophyte cover occurred in both the overall kōura and eel models, but was significantly positively correlated with eels and significantly negatively associated with both total and YOY kōura, suggesting it may be a habitat where kōura are vulnerable to predation. This result is in contrast to the findings of Parkyn (2000) who established kōura were positively associated with cover dominated by macrophytes in pasture streams. However, the paucity of kōura at these sites in the present study (where there were only a small number of sites with macrophyte cover) could also have been due to other factors such as unsuitability of habitat, finer substrates and/or flow variability.

While the results of this study generally support part of Hypothesis 1 (see section 5.1) through the inverse relationship detected between kōura density and eel density, the inclusion of both total and small eels in the kōura model and exclusion of eels >300 mm does not support the role of larger eels as dominant predators regulating kōura densities. Rather, there seems to be an interaction between eel size and refugia availability which, coupled with larger-scale factors, influenced kōura densities across sites.

The addition of bullies to the overall kōura density model increased its explanatory power by 1% and there was no correlation between total kōura density and bullies, or between bullies with YOY kōura. However, at some sites bullies dominated fish catches and their density was positively correlated to total fish density which, in turn, was significantly and inversely correlated to both total and YOY kōura densities. This may suggest that bullies act in concert with small eels and may, at least in part, contribute to the observed negative correlations between total fish and kōura densities. All three bully species have similar diets consisting of stream insects, small fish, crustaceans and snails (McDowall, 2000), which coincide with the preference of invertebrate food by small kōura (Parkyn, 2000) indicating competition for food may occur at high densities. Although bullies are unlikely to exert predation pressure on larger kōura, very small YOY kōura may be vulnerable to predation. Bullies may not have featured in the DistLM model selection due low variability between sites compared to other model variables selected. Findings in the current study weakly support Hypothesis 2 (see section 5.1), that the density of *Gobiomorphus* spp. will influence only YOY kōura density, as part of a broader benthic invertivorous guild of fish that prey on small kōura.

As previously discussed in Chapter 4, findings from this study indicate a wide range of environmental variables influence kōura density which operate at both catchment and reach-scales. Although fish communities no doubt exert some degree of influence over kōura populations, establishing the mechanisms driving such relationships is difficult given the range of potential factors influencing kōura and fish densities and their distribution. Although there was some indication in this study that eels and small invertivorous fish could influence kōura densities during summer in these hill-country streams, it seems likely that where suitable habitat is available, in particular adequate stable refugia, then kōura and the native fish community experiencing natural levels of recruitment (i.e., not artificially stocked) are able to co-exist.

5.4.3 Environmental factors affecting eels

Distance to sea was hypothesised to be the dominant factor influencing eel density differences between sites with comparatively minimal influence from instream habitat or local-scale riparian planting age factors (Hypothesis 3; see section 5.1). Findings in the current study strongly support this hypothesis with the majority of eels and other diadromous fish found in catchments closer to the sea or at lower elevations, as found in previous studies (e.g., Joy & Death 2001 and Hayes *et al*. 1989). Thus, distance downstream to the sea was significantly negatively correlated with densities of total eel and eels ≤300 mm, and appeared in both the upstream catchment model and the overall model for total eel density. Generally, eel densities, particularly eels ≤300 mm were higher at sites <15 km from the sea, which were primarily located in the Whaingaroa site spatial cluster. The exception was TU1 (located 128 km inland), located in the Whatawhata cluster, where higher densities of eels ≤300 mm were likely related to the low site elevation (60 m asl) and downstream gradient enabling relatively unimpeded access up the Waikato and Waipā rivers. Almost all eels >300 mm were longfin and were generally associated with sites further inland. Although not specifically measured in this study, field observations support the premise that elevation is also an important driver in eel species distribution at these study sites. These distribution patterns support existing knowledge of the ability and migratory drive of longfin eels to penetrate further inland and infiltrate higher elevation sites than shortfin eels (McDowall, 1990).

The appearance of riparian buffer extent and percentage grass in both the upstream catchment and overall models for total eel density appeared to reflect a negative correlation between eels and upstream riparian buffer vegetation and a positive relationship between eel density and upstream pasture. Indeed, many of the sites with the highest densities of eels had upstream catchments dominated by pasture with variable riparian buffer vegetation cover. Conversely, many sites around the flanks of Pirongia had minimal upstream pasture, native forested headwaters and subsequent low densities of eels associated with distance inland and elevation.

The links between catchment land-cover and fish density and abundance have previously been studied (e.g., Jowett *et al* (1996) and Hayes *et al* (1989)). Both studies indicated that fish density was more likely to reflect distance downstream from the sea rather than land-cover influencing habitat preferences. Findings in the present study appear to support these previous studies, with distance downstream to the sea a major influence over fish density compared to land-cover.

Instream habitat variation featured strongly in the overall eel density model. Four variables associated with hydraulic conditions were represented along with two instream cover variables. The presence of gravel, silt, riffle and run variables illustrates the wide variability of habitat utilised by eels, and potentially reflects differences in habitat use by eels ≤300 mm and those >300 mm as reported by Glova *et al.* (1998) for three coastal New Zealand streams. That study found longfin eels <300 mm were associated with a range of velocities and substrate sizes, shortfin eels <300 mm were evenly spread across all habitats, while eels of ≥300 mm of both species showed no consistent pattern of habitat use across the streams sampled. Hayes *et al.* (1989) studied fish assemblages in the Mokau River and established relationships of both adult and juvenile longfin eels with finer substrates associated with pools, as well as riffles and coarse substrates at moderate depths. This study indicated positive associations of eels with run habitat, silt substrate and macrophyte cover and indicated negative associations with riffles and gravel substrate, supporting previous studies indicating the wide range of substrate types, velocity regimes and depths utilised by both eel species.

The importance of riparian cover for eels has previously been reported in Glova (1999). Although riparian factors accounted for 38% of eel density in the riparian model, only two riparian variables featured in the overall eel density model. The inclusion of number of saplings and percentage bare ground reflects the variability of eel density across a range of riparian planting ages and native sites. Higher numbers of saplings were often found at old riparian aged sites, although saplings were also common at younger sites with remnant native vegetation and numbers varied at native forest sites depending on vegetation structure and species composition. Bare ground was common in heavily shaded sites indicative of native and older riparian plantings, but was also found in younger sites where planting density was high.

Eels have previously been described by Jellyman (1989) as the most widespread and frequently encountered native freshwater fish in New Zealand and can be found in almost all habitats with access to the sea. Glova *et al.* (1998) described eels belonging to the *Anguilla* genus as having ecological plasticity reflected in their widespread distribution and ability to exist in a diverse range of habitats. These characteristics make it difficult to attribute specific instream or riparian habitat variables, or catchment-scale influences, to driving eel densities. When the overarching influence of distance downstream to the sea and elevation were also considered, there appear to be multiple influences regulating eel distribution and density at a reach-scale. Nevertheless, the current study did highlight some potentially important instream habitat factors (root complexes, gravel and silt substrate, riffle and run habitat and macrophyte cover) and riparian/upstream catchment variables (grass vegetation, buffer in woody vegetation, distance downstream to the sea, number of saplings, and bare ground) influencing eel density in small hill-country streams with good access to the sea.

5.4.4 Effects of riparian planting age on fish

The hypothesis that fish density would not be influenced by riparian planting age, because of the over-riding influence of distance to sea for diadromous species and the known affinity of eels for open sites was confirmed in this study. Thus, there were no statistically significant differences found between total fish, total bullies or eel population characteristics and riparian planting age groups for sites in this study, indicating riparian planting age per se. exerted minimal influence on fish densities. Similarly, no significant differences were found for variation in fish characteristics between older riparian sites and native forest sites confirming the second part of Hypothesis 4 (see Section 5.1). Indeed, a range of riparian management factors other than plant age moderate the influence of riparian vegetation on streams. These include factors such as planting density, planting proximity to the stream bank, plant species used and extent of remnant native and exotic vegetation present in the riparian zone.

The influence of riparian cover on fish has been addressed in previous studies which appear to support the findings of this current study. Jowett and Richardson (1995) surveyed 34 New Zealand rivers and concluded that riparian vegetation and instream cover did not appear to have over-riding influences on fish communities. Holmes *et al.* (2016) attempted to establish links between riparian fencing and eel and brown trout density in the Waikakahi catchment expecting to find high habitat quality such as cover provided by riparian vegetation, relatively deep water and low sediment inputs would provide greater fish habitat and therefore greater fish density, however, this proved not to be the case for both species of eel. However, in surveys of three coastal streams, Glova *et al.* (1998) surmised that both species of eel >300 mm length were strongly associated with cover, with larger longfin eels utilising a variety of cover types such as macrophyte cover, banks, and instream debris while shortfin eels more closely associated directly with riparian cover. In the current study, it appears that although there were large variations in fish characteristics between different-aged riparian planting groups, the overall variation in fish and specifically eel density between riparian age classes was more closely related to the overarching influences of distance to sea, elevation and gradient rather than influence of riparian planting age.

Findings in Chapter 3 indicated that there no differences in key instream habitat variables between older riparian plantings and native forest sites. This was further confirmed in Chapter 4 for total kōura density and in this chapter for a range of descriptors of fish community composition. This finding could be interpreted as older riparian plantings sites acting similar to native forested sites in terms of providing similar habitat conditions suitable for kōura and fish communities. Indeed, it was perhaps not surprising that no significant differences for fish densities were established between older riparian and native forest sites, or among different riparian plant age groups, given the strong influence of geographic factors such as distance downstream to the sea structuring fish community and density, and upstream catchment characteristics in structuring habitat. For eels, at least the ability to utilise a wide range of habitats (Glova *et al.*, 1998) limits the ability to establish strong links between different types of sites based in reach-scale riparian characteristics.

Chapter 6

General discussion

6.1 Study purpose and aim

Riparian fencing and planting are commonly used in New Zealand to mitigate pastoral land-use impacts on streams (McKergow *et al.*, 2016), with the primary aim to achieve improvements in water quality (Ministry for the Environment, 2001). Instream habitat is often the forgotten dimension when addressing ecosystem health concerns, but it is a key factor along with water quality affecting native aquatic species abundances. The effects of riparian fencing and planting on improving instream habitat have been explored previously (e.g., Greenwood et al. 2012), however, findings have often been equivocal or context-dependent. Most studies evaluating riparian planting influences on stream ecosystems have focused on structural components such as shade (Quinn *et al.*, 1997) and macroinvertebrate metrics (Jowett *et al.*, 2009; Quinn *et al.*, 2009).

The freshwater crayfish *Paranephrops planifrons*, or kōura, is usually not specifically targeted as a measure of stream restoration success, partially because they are generally underestimated using conventional stream sampling methods utilised for fish and other macroinvertebrates (Collier *et al.*, 1997). Kōura are an important component of New Zealand stream ecosystems (Parkyn *et al.*, 2001), and are valued by Māori as a cultural keystone species and source of food (Noble *et al.*, 2016). Previous studies have established differences between kōura populations in pasture and native streams (Hicks & McCaughan, 1997; Parkyn, 2000), however, the influence of riparian management, specifically planting age, on kōura distribution and abundance in pastoral streams in the western Waikato has not previously been tested.

The overarching aim of this thesis was to provide key information for environmental managers and kaitiaki on the response of kōura populations to riparian fencing and plantings at different stages of growth along pastoral streams. To establish the key drivers influencing kōura density, I firstly investigated linkages between upstream catchment and riparian characteristics with instream habitat variables measured at the reach-scale. Secondly, I explored the relative influences of upstream catchment, riparian and instream habitat variables on kōura abundance and size structure, and assessed influence of riparian planting age on kōura. Finally, I identified associations between fish communities and kōura populations within sample reaches in relation to upstream catchment, instream habitat and riparian factors, with a particular emphasis on eels (*Anguilla* spp.) as a potential key predator. The results provide insights into important variables associated with kōura abundance and recruitment that can guide future riparian management of hill-country streams aimed at enhancing kōura populations.

6.2 Methodological approach

I used a 'space for time' approach to evaluate the effects of riparian vegetation age on hill-country streams that were planted at different times across three spatial clusters, each with its own pastoral and native site comparison. To achieve this, I interrogated Waikato Regional Council (WRC) databases for potential sites that fitted specific criteria, and utilised the New Zealand Freshwater Fish Database and WRC fish monitoring database to assess the likelihood of kōura presence. Potential sites were evaluated with WRC staff and landowners in conjunction with site reconnaissance.

GIS was used to derive relevant upstream catchment-scale and reach-scale variables for each study site using an existing digital elevation model (8 m resolution). This exercise, coupled with additional available spatial information, was utilised to establish upstream catchment boundaries, drainage network length, upstream extent of native riparian vegetation (20 m buffer), distance to native forest headwaters, upstream land-cover and geology, and site and upstream catchment gradients. Riparian plots were surveyed at each site to establish vegetation structure and composition to link with riparian age, something that is often overlooked in studies of riparian effects on stream ecosystems. In addition, comprehensive instream habitat measurements were collected which specifically targeted expected kōura habitat and parameters likely to be influenced by riparian condition.

A novel sampling approach was developed to optimise kōura and fish capture utilising a combination of electric-fishing, substrate disturbance and induced flow by sweeping with soft-bristled brooms. Two passes of multiple 10-m sub-reaches were conducted to provide a standard measure of abundance between sites. This method provided a sampling tool that limited damage to crayfish, captured a range of kōura and fish sizes, and can be applied more broadly by agencies conducting standardised kōura abundance surveys.

6.3 Limitations and constraints

A range of limitations and constraints were encountered during this research. Firstly, site selection was based on a set of criteria designed to reduce variability in key environmental drivers between sites (e.g., stream size) and accommodate electric-fishing as a sampling tool. Selection parameters included length and age of riparian planting, channel width and depth, stream order, and geographic location. Although attempts were made to minimise variation to tease out the effects of riparian planting age, site selection was limited due to the paucity of similar sites which met the study criteria. Implementation of riparian management on pastoral farms has often been incentivised based on priority catchments such as those identified in WRC work programmes (e.g., Waikato Regional Council 2014), as well as the willingness of landowners to undertake works. This approach can result in either unconnected riparian sites spread across landscapes or catchments, or conversely, sites located in small geographic areas across multiple landowners. Additionally, potential sites often had small-scale plantings below the minimum selection threshold of 200 m length, or included only fencing but no planting which was unsuitable for inclusion in this study. Although, some catchments contained multiple sites, care was taken to maintain some independence by ensuring they were >500 m apart or were on different tributaries. Effort was also made to spread sites around the three spatial clusters available. Overall, despite best efforts, the sites sampled varied greatly so that isolating the effect of riparian age was challenging.

While the ‘space for time’ approach enabled assessment of a large number of sites with different-aged riparian planting within a limited period, it imposed some additional limitations on interpreting the influence of riparian planting age. A wide range of riparian management techniques were used among sites, meaning each site was managed differently depending on the objectives of the landowner, industry requirements and/or the desired outcomes of the riparian management. Key parameters that varied between riparian sites included width of buffers, type of vegetation planted, density of plantings and extent of remnant native and exotic vegetation. These combined with varying valley slopes, all limited comparisons between either, sites of the same age category, or groups of sites with different ages. Additionally, high variability in upstream catchment factors appeared to exert strong influences on instream habitat that over-rode riparian planting age influences. Amongst these factors was the proximity of many riparian and pasture sites to the moderating influence of upstream native forest headwaters which created a downstream ‘halo’ of higher habitat and water quality than would have occurred in fully pastoral catchments, potentially limiting the responses of some streams to riparian plant growth.

Physicochemical parameters have previously been reported to influence crayfish presence and abundance in streams (e.g., Olsson *et al.* 2006), but their characterisation requires regular sampling to capture daily and seasonal variability. In this study, all measurements were one-off and made mostly during the early morning when water temperatures would be low due to night-time cooling, and dissolved oxygen is typically low due to night-time respiration. Although these spot measurements did provide additional information for characterising study sites, they were limited in their ability to identify potentially important influences on kōura populations which occur over larger timescales. For this reason they were used in only in models predicting differences in kōura density among sites, and were not analysed in relation to riparian planting age.

Establishing relationships between fish and kōura at a 50-m reach scale is constrained by fish being more mobile than kōura, limiting inferences that can be made between the fish community and its potential influence on kōura. The chosen sampling reach may not have contained suitable habitat for some fish species during the day-time, however, such habitat may have existed in close proximity enabling fish to still influence the sample reach during upstream and downstream movements. Also, the conclusions made reflect summer day-time instream habitat use by fish and kōura which are both known to be more active at night. While existing protocols (e.g., David *et al.* (2010)) illustrate the need to sample 150 m reaches to capture >90% of the fish species present with one-pass electric-fishing, the present study placed more emphasis on fishing a shorter reach more intensively (two passes) to acquire abundance data given the primary target was kōura. Associations found between kōura and fish communities in this study therefore reflect day-time interactions with the dominant fish species in Waikato hill-country streams.

6.4 Key findings

Frissell *et al.* (1986) and Winterbourn (1981) presented the concept of stream catchments consisting of a hierarchy of nested spatial levels, each constraining the level below to influence the structure of biotic communities. This study has examined upstream catchment-scale influences as well as reach-scale instream habitat and riparian characteristics to establish key drivers for influencing kōura abundance, and has established findings which generally align with the hierarchy concept.

Upstream catchment factors were the key drivers for instream habitat variability between sites. Dominant factors were upstream catchment size and geology followed by various classes of upstream land-cover and distance to native forest, all influencing stream channel morphology, substrate, velocity and cover. Reach-scale riparian factors had little bearing on instream habitat, and although stream bank slope featured consistently as an influence, it is arguably more closely linked with catchment-scale factors such as topography and gradient than riparian characteristics.

While kōura abundance was influenced by some upstream catchment factors, particularly catchment size, gradient, geology and various land-cover classes, they were most strongly influenced by reach-scale instream habitat factors such as substrate type and hydraulic habitat, as well as organic components such as macrophyte cover and root complexes. Riparian reach-scale factors did not feature strongly in models, however, stream bank slope and sapling density did explain some of the variation in kōura densities between sites, and several important habitat features were linked to riparian vegetation function (e.g., root complexes). Important instream habitat variables for young-of-the-year (YOY) kōura differed from total kōura abundance. Notably, YOY densities were positively associated weakly with cover by cobble and sand substrates and strongly negatively with gravel. Root complexes were also important for YOY kōura, however, both total and YOY kōura, were negatively associated with macrophyte cover.

Diadromous fish abundance was overwhelmingly influenced by downstream distance to the sea. The two most abundant fish types captured were eel and bully species. Eel catches were dominated by longfin eels with eel density (particularly eels ≤300 mm) higher at sites close to the coast or at lower elevations. Instream habitat appeared to influence eel density variation between sites, primarily reflecting the wide habitat use of eels and differences in habitat types utilised by eels ≤300 mm and those >300 mm. With the exception of downstream distance to the sea, both upstream catchment and riparian variables were secondary to instream habitat factors in terms of their modelled influences on eel densities.

There was some evidence to support a negative interaction between kōura and eels and/or bullies as moderators of total and YOY kōura densities. Total fish density exerted influences in models of both total and YOY kōura densities, driven mainly by densities of eels ≤300 mm and bullies, largely reflecting their abundance at sites closer to the coast or at low elevation. Unexpectedly, eels >300 mm did not feature as a strong influence on either total or YOY kōura.

Strong overlap between eel and kōura models occurred with macrophyte cover and root complexes. Both total and YOY kōura were strongly correlated with root complexes and negatively with macrophyte cover. Conversely, eels were negatively associated with root complexes and positively to macrophyte cover. These contrasting responses perhaps indicate high susceptibility of kōura to predation in macrophytes and root complexes providing refugia from predation. This finding underscores the direct and indirect effects that riparian vegetation can potentially have on kōura by providing shade to limit macrophyte growth and instream habitat in the form of submerged root complexes. Shade levels did follow the expected trajectory of increasing with planting age in the current study.

The interacting relationships between the hierarchical factors explored in this study and their relative influences on kōura and fish are summarised in Figure 6.1.

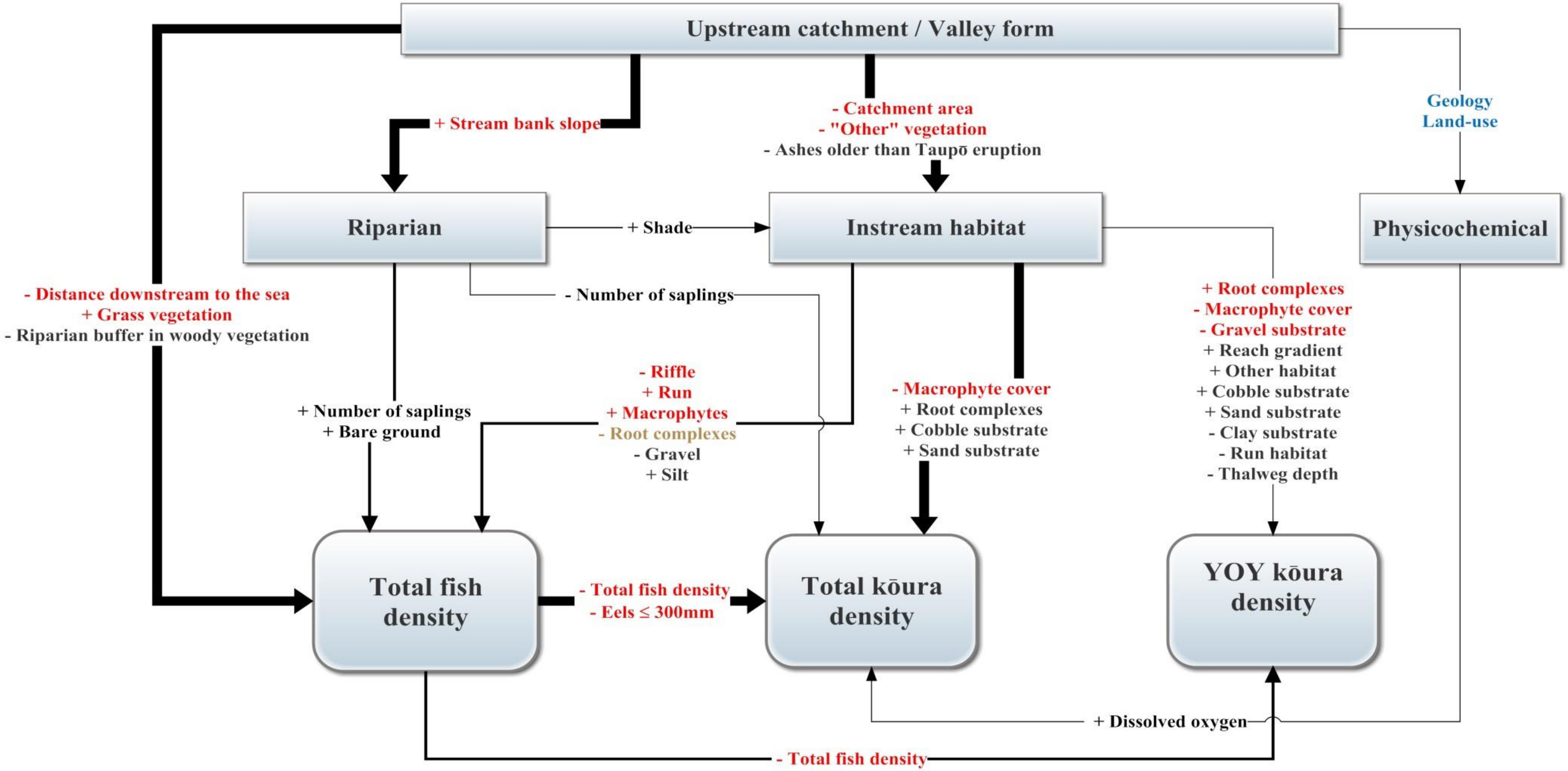

Figure 6.1: Summary diagram of relationships between total kōura density, young-of-the-year (YOY) kōura density and total fish density with upstream catchment, instream habitat, riparian and physicochemical variables. For the "Total fish density" box, upstream catchment, riparian and instream drivers were for total eel density only. Red = r_s = >0.7 and/or significantly correlated, Brown = r_s 0.4-0.7, black = r_s <0.4. Blue = known influences not tested in this study. Arrow size indicates strength of relationships: thick = strong, medium = moderate and thin = weak.

6.5 Conclusions and implications for management

Riparian planting age *per. se* appeared to have less influence on kōura abundance than upstream catchment-scale influences and instream habitat, however, some important instream habitat characteristics were associated with riparian vegetation. Several previous studies (e.g., Parkyn *et al*. 2003) have illustrated the difficulty in assessing the benefits of riparian planting and fencing on stream ecosystems, however, there seems little doubt that the fundamental principle of excluding stock from streams and revegetation of riparian margins will benefit stream ecosystems in multiple ways. However, direct benefits to kōura density appear to be dependent on a wide range of other factors that relate to site and upstream catchment characteristics as well as riparian management practices (Figure 6.1).

Kōura were clearly able to live in a wide range of stream environments associated with varying land-cover and levels of riparian management, and to co-exist with potential native predators where instream habitat conditions, particularly availability of root complex refugia, were suitable. This and other studies indicate important considerations for achieving positive outcomes from riparian management include: i) the stream's position within the catchment, including size and upstream geomorphology and land-use (Burrell *et al.*, 2014); ii) riparian buffer length; iii) the spatial arrangement of riparian management sites within the catchment; and iv) proximity to colonists, particularly source populations in native forest headwaters (Parkyn *et al.*, 2003; Parkyn & Collier, 2004).

Identifying sites for kōura enhancement, may be as simple as firstly ensuring upstream catchment influences provide suitable reach-scale habitat conditions. Habitat can then be enhanced through riparian planting of trees that, over time, will produce root complexes extending into the water and also provide complexity along stream banks. Mahoe (*Melicytus ramiflorus*) and tree ferns (*Cyathea* spp. and *Dicksonia* spp.) appeared to be especially effective at achieving this based on field observations.

To achieve kōura enhancement in streams using riparian management, efforts would best be focused on streams with the following attributes:

- Intermediate-reaches that are not coastal due to the potential influence of high populations of eels and other invertivores, but have some connectivity to native forest headwaters providing stable habitat and a source of kōura for downstream colonisation;
- Streams which contain existing suitable kōura habitat that can be further enhanced;
- Streams draining smaller catchments with areas of low-medium velocity, and low flood variability;
- Sites with areas of suitable substrates to provide cover such as cobble and silt/sand areas for YOY kōura;
- Sites which have existing remnant riparian vegetation providing some instream habitat that could be further enhanced by riparian planting;
- Catchments that contain connected areas of riparian retirement linking upstream headwaters to lowland reaches.

Limited gains for kōura through riparian planting are likely in larger or degraded streams or in the presence of additional stressors such as large populations of native or introduced fish predators where existing refugia and stable substrates are lacking.

Ultimately, riparian management could be used as a tool to enhance kōura populations in suitable streams. Indeed, increased kōura abundance and persistence in managed streams could be an additional metric to measure positive outcomes from riparian management at appropriate locations. The role of kōura as both ecological engineers and cultural keystone species, and their relatively sessile nature lend themselves to potentially being used as an indicator species to provide an important measure of management outcomes on overall stream health.

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Appendices

Appendix 1: Upstream catchment characteristics of 26 study sites in Pirongia, Whaingaroa and Whatawhata. Upstream distance (m) = distance from upstream end of sample reach to delineated >20 ha catchments.

Site	Cluster	Spatial cluster	Riparian planting age group	Catchment area (ha)	Upstream distance (m)	% buffer in woody vegetation	Grass vegetation (%)	Indigenous vegetation (%)	Scrub vegetation (%)	Other vegetation (%)	Gradient change (m/m)	Ashes older than Taupō eruption (%)	Volcanic rock (%)	Greywacke (%)	Sandstone (%)	Mudstone (%)	"Other" geology (%)	Distance native forest (m)
MK1	PRG	Pasture	0	597	9823	84.24	15	84	0	1	0.06	9.37	82.36	0	0	0	8.27	2462
R1	PRG	Recent	2	427	8990	79.62	27	73	0	1	0.07	14.44	85.56	0	0	0	0	3135
TP3	PRG	Recent	2	30	532	17.32	73	27	0	0	0.12	0	100	0	0	0	0	N/A
KA1	PRG	Recent	3	484	7704	24.69	64	26	8	1	0.01	11.93	0	31.93	41.85	14.28	0	2548
TP5	PRG	Recent	4	27	395	35.71	68	32	0	0	0.10	93.13	6.87	0	0	0	0	356
TP6	PRG	Recent	4	368	7903	75.76	52	48	0	0	0.05	1.85	98.15	0	0	0	0	3287
MM1	PRG	Mid	6	190	5978	60.14	30	70	0	0	0.07	12.46	87.54	0	0	0	0	1552
TP2	PRG	Mid	8	22	69	60.00	58	42	0	0	0.09	0	100	0	0	0	0	N/A
O1	PRG	Mid	9	43	289	94.00	100	0	0	0	0.01	63.16	36.84	0	0	0	0	N/A
TP1	PRG	Mid-late	12	78	1685	82.46	17	83	0	0	0.14	0	100	0	0	0	0	686
R2	PRG	Native	N/A	37	747	100	0	100	0	0	0.15	0	100	0	0	0	0	0
T1	WG	Pasture	0	80	1754	39.02	43	57	0	0	0.12	12.58	87.42	0	0	0	0	1320
BC2	WG	Recent	5	111	1518	28.74	47	11	42	0	0.02	12.50	87.50	0	0	0	0	332
BC1	WG	Mid-late	11	37	493	8.64	94	1	5	0	0.06	35.53	64.47	0	0	0	0	N/A
TA1	WG	Mid-late	12	270	6267	37.35	73	25	0	1	0.05	21.07	78.93	0	0	0	0	4663
AA1	WG	Mid-late	13	223	5422	65.83	77	23	0	0	0.06	11.49	88.51	0	0	0	0	4170
OM1	WG	Mid-late	14	89	712	86.19	60	36	0	4	0.03	21.26	78.74	0	0	0	0	N/A

Site	Cluster	Spatial cluster	Riparian planting age group	Catchment area (ha)	Upstream distance (m)	% buffer in woody vegetation	Grass vegetation (%)	Indigenous vegetation (%)	Scrub vegetation (%)	Other vegetation (%)	Gradient change (m/m)	Ashes older than Taupō eruption (%)	Volcanic rock (%)	Greywacke (%)	Sandstone (%)	Mudstone (%)	"Other" geology (%)	Distance native forest (m)
TM1	WG	Old	16	329	8061	47.99	81	19	0	0	0.02	1.80	98.20	0	0	0	0	N/A
W1	WG	Old	23	398	7936	63.74	26	72	0	2	0.06	7.70	82.60	0	0	0	9.70	2052
W2	WG	Native	N/A	317	5319	91.07	19	80	0	1	0.08	2.21	97.55	0	0	0	0.24	0
K3	WW	Pasture	0	256	4081	86.38	29	62	2	7	0.03	0	0	100	0	0	0	855
TU1	WW	Recent	1	179	2858	75.37	24	21	0	55	0.02	0	0	100	0	0	0	1166
TU2	WW	Recent	5	794	12784	46.93	61	11	1	27	0.01	0	0	60.78	27.79	2.54	8.89	5340
K1	WW	Old	18	18	0	97.23	59	41	0	0	N/A	0	0	99.66	0.34	0	0	N/A
K2	WW	Old	18	240	4123	78.18	35	1	0	64	0.03	0.01	0	93.30	0	6.69	0	N/A
WK1	WW	Native	N/A	318	4081	100	0	99	0	1	0.02	0	0	100	0	0	0	0

Appendix 2: Riparian characteristics of 26 study sites in Pirongia, Whaingaroa and Whatawhata.

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Avg. Stream bank slope (°)	Avg. buffer width (m)	Number saplings (per m^2)	Basal area (m^2 per 20m^2)	Bare Ground (%)	Mosses/Liverworts (%)	Leaf Litter (%)	Vegetation ≤ 0.3 m (%)	Vegetation > 0.3 m (%)	Shrubs (%)	% Shrubs/saplings native	% shrubs/saplings exotic
MK1	PRG	Pasture	0	8.50	0	0.01	0	5.00	0	0	93.33	0	1.67	0	100
R1	PRG	Recent	2	11.67	6.13	0	0.09	0	0	0.83	0	75.83	23.33	76.67	23.33
TP3	PRG	Recent	2	18.17	8.26	0.03	0.01	0	0	0	0	79.17	20.83	100	0
KA1	PRG	Recent	3	28.33	6.18	0.18	0	4.17	0	0	0	45.00	50	83.33	16.67
TP5	PRG	Recent	4	21.67	10.81	0.17	0.07	0	0	0.83	0	45.83	53.33	98.33	1.67
TP6	PRG	Recent	4	12.83	3.80	0.09	0.01	0	0	0	1.67	70.00	28.33	96.67	3.33
MM1	PRG	Mid	6	24.33	7.18	0.13	0.12	0	0	6.67	0	46.67	46.67	91.67	8.33
TP2	PRG	Mid	8	20.67	5.20	0.01	0.05	0	0	0	0	65.00	35	100	0
O1	PRG	Mid	9	18.50	11.88	0.11	0.09	6.67	21.67	18.33	0	0	53.33	68.33	31.67
TP1	PRG	Mid-late	12	26.33	6.70	0.22	0.17	10.00	6.67	14.17	0	38.33	30.83	100	0
R2	PRG	Native	N/A	28.00	N/A	0.10	0.12	3.33	6.67	41.67	3.33	10.00	35	100	0
T1	WG	Pasture	0	11.17	0	0	0	12.50	0	0	83.33	0	4.17	0	0
BC2	WG	Recent	5	12.50	≥20.00	0.04	0.03	0	0	0	0	75.00	25	50.00	16.67
BC1	WG	Mid-late	11	26.67	14.90	0.53	0.02	5.00	0	12.50	0.83	51.67	30	50.00	50.00
TA1	WG	Mid-late	12	13.00	16.80	0.20	0.12	4.17	0.83	12.50	3.33	43.33	35.83	98.33	1.67
AA1	WG	Mid-late	13	21.00	≥20.00	0.28	0.06	0	0	5.83	0	60.00	34.17	92.50	7.50
OM1	WG	Mid-late	14	6.33	≥20.00	0.38	0.10	8.33	5.83	10.83	0	5.00	70	98.33	1.67
TM1	WG	Old	16	13.83	7.70	0.31	0.02	0.83	0.83	1.67	0	30.83	65.83	98.33	1.67
W1	WG	Old	23	14.00	≥20.00	0.65	0.23	5	5	7.5	0	55.00	27.50	91.67	8.33
W2	WG	Native	N/A	12.33	N/A	0.40	0.21	13.33	6.67	20.00	0	16.67	43.33	98.33	1.67

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Avg. Stream bank slope (°)	Avg. buffer width (m)	Number saplings (per m^2)	Basal area (m^2 per 20m^2)	Bare Ground (%)	Mosses/Liverworts (%)	Leaf Litter (%)	Vegetation ≤ 0.3 m (%)	Vegetation > 0.3 m (%)	Shrubs (%)	% Shrubs/saplings native	% shrubs/saplings exotic
K3	WW	Pasture	0	38.33	0	0.06	0.09	3.33	0	0.83	30.00	60.83	5.00	58.33	8.33
TU1	WW	Recent	1	16.00	≥20.00	0	0.01	5.00	0	3.33	0	81.67	10.00	100	0
TU2	WW	Recent	5	33.33	4.43	0.25	0.06	0	0	0	0	87.50	12.50	95.00	5.00
K1	WW	Old	18	35.17	≥20.00	0.64	0.16	10.00	4.17	55.83	0	0	30.00	100	0
K2	WW	Old	18	27.83	≥20.00	0.55	0.14	1.67	2.50	20.83	11.67	33.33	30.00	90.00	10.00
WK1	WW	Native	N/A	16.33	N/A	0.39	0.09	0	30.00	25.83	0	0	44.17	100	0

Appendix 3: Instream habitat characteristics of 26 study sites in Pirongia, Whaingaroa and Whatawhata.

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Channel width (m)	Wetted width (m)	Thalweg depth (m)	Shade (% canopy cover)	CPOM (m^2)	Wood (m^2)	Root complexes (m^2)	Undercuts (m)	Overhanging vegetation (%)	Bedrock (%)	Boulder (%)	Cobble (%)	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Run habitat (%)	Riffle habitat (%)	Pool habitat (%)	"Other" habitat (%)	Macrophyte cover (%)	Reach gradient (m/m)
MK1	PRG	Pasture	0	5.14	3.42	0.25	17.68	0.01	0.01	0.30	15.50	6.50	0	14	48	34	2	2	0	40	50	10	0	0.40	0.02
R1	PRG	Recent	2	5.48	4.48	0.25	44.72	0.01	0.01	2.42	23.59	13.00	0	12	38	44	6	0	0	40	60	0	0	0	0.03
TP3	PRG	Recent	2	1.22	0.77	0.14	55.07	0.08	0.01	5.91	29.84	80.00	0	32	16	20	16	16	0	54	33	11	2	0	0.09
KA1	PRG	Recent	3	2.27	2.07	0.23	50.08	0.01	0.02	3.18	27.40	55.00	0	2	0	70	16	12	0	80	16	4	0	0.40	0.01
TP5	PRG	Recent	4	1.26	0.76	0.16	28.91	0.06	0.01	0.98	12.43	44.50	0	6	2	22	10	48	12	100	0	0	0	0	0.01
TP6	PRG	Recent	4	3.96	3.82	0.30	17.06	0.01	0	1.17	38.81	32.00	0	22	30	28	20	0	0	52	48	0	0	0.20	0.05
MM1	PRG	Mid	6	3.40	3.10	0.36	77.38	0.03	0	3.14	16.70	78.00	0	20	42	32	4	2	0	44	52	4	0	0	0.03
TP2	PRG	Mid	8	1.56	1.14	0.16	77.48	0.14	0	9.16	1.66	36.00	0	10	24	16	8	8	34	44	54	2	0	0	0.22
O1	PRG	Mid	9	1.33	1.18	0.18	96.67	0.05	0	1.27	8.02	38.00	0	22	32	36	6	4	0	35	35	29	1	0	0.17
TP1	PRG	Mid-late	12	2.35	1.64	0.27	78.10	0.06	0.01	25.48	13.35	52.50	0	26	16	2	20	36	0	39	46	6	9	0	0.09
R2	PRG	Native	N/A	2.16	1.31	0.14	95.26	0.11	0.02	4.39	2.52	46.00	0	10	28	38	18	6	0	40	48	6	6	0	0.18
T1	WG	Pasture	0	1.70	1.60	0.33	0.68	0	0	0	44.56	12.50	0	6	34	28	16	16	0	32	60	8	0	1.00	0.07
BC2	WG	Recent	5	1.14	1.11	0.14	5.72	0	0	0	16.50	100	2	10	26	22	22	18	0	89	11	0	0	26.00	0.04
BC1	WG	Mid-late	11	1.96	1.40	0.11	80.81	0.02	0	3.94	12.65	55.00	28	22	12	16	6	14	2	31	24	24	21	0	0.21
TA1	WG	Mid-late	12	2.18	1.94	0.30	61.93	0.02	0.01	0.39	15.70	45.00	0	18	28	42	6	6	0	70	28	2	0	0	0.01
AA1	WG	Mid-late	13	2.42	2.02	0.23	44.41	4.75	1.97	0.29	25.50	78.00	0	18	26	30	12	14	0	42	42	16	0	0	0.04
OM1	WG	Mid-late	14	1.44	1.24	0.44	90.58	0.02	0.03	29.66	22.75	89.00	0	0	0	14	16	70	0	93	2	5	0	2.00	0.01
TM1	WG	Old	16	2.29	1.70	0.37	27.20	0.02	0	3.58	13.25	100	0	38	30	30	0	2	0	76	16	8	0	2.80	0.01
W1	WG	Old	23	3.14	2.77	0.26	96.46	0.03	0.01	6.20	17.75	48.00	0	0	32	40	18	4	6	62	24	14	0	0	0.02
W2	WG	Native	N/A	3.81	3.31	0.27	94.85	0.01	0.01	4.04	20.15	19.50	0	22	30	38	10	0	0	60	36	4	0	0	0.04

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Channel width (m)	Wetted width (m)	Thalweg depth (m)	Shade (% canopy cover)	CPOM (m^2)	Wood (m^2)	Root complexes (m^2)	Undercuts (m)	Overhanging vegetation (%)	Bedrock (%)	Boulder (%)	Cobble (%)	Gravel (%)	Sand (%)	Silt (%)	Clay (%)	Run habitat (%)	Riffle habitat (%)	Pool habitat (%)	"Other" habitat (%)	Macrophyte cover (%)	Reach gradient (m/m)
K3	WW	Pasture	0	2.35	2.09	0.31	53.30	0.00	0	2.25	22.64	18.00	0	0	24	62	14	0	0	90	0	10	0	0	0.01
TU1	WW	Recent	1	2.65	1.52	0.14	46.70	0.02	0.03	0.23	18.30	69.00	0	0	10	68	20	2	0	85	13	2	0	13.80	0.07
TU2	WW	Recent	5	2.66	2.27	0.30	43.78	0.01	0.01	0.03	4.50	19.50	0	0	0	40	32	16	12	100	0	0	0	2.20	0.04
K1	WW	Old	18	1.46	1.36	0.21	96.82	0.05	0.02	5.65	27.90	12.00	6	0	0	20	28	46	0	77	7	16	0	0	0.04
K2	WW	Old	18	2.57	2.27	0.13	85.85	0.02	0.01	1.58	21.00	40.00	22	0	8	38	14	18	0	68	14	16	2	0	0.02
WK1	WW	Native	N/A	3.53	3.63	0.34	92.92	0.02	0.01	3.90	43.80	4.50	0	10	14	46	24	6	0	48	32	20	0	0	0.02

Appendix 4: Marginal tests for DistLM analysis for upstream catchment and riparian variables for instream habitat dissimilarity matrix for 26 study sites throughout Pirongia, Whaingaroa and Whatawhata.

Upstream catchment					Riparian				
Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.	Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.
Catchment area (ha)	62.73	3.26	0.001	0.12	Number of saplings per m^2	39.74	1.96	0.026	0.08
Volcanic rock (%)	61.35	3.18	0.002	0.12	Leaf litter (%)	38.54	1.90	0.036	0.07
Gradient change (m/m)	41.26	2.05	0.013	0.07	Average buffer width (m)	30.70	1.49	0.124	0.06
“Other” vegetation (%)	37.95	1.87	0.051	0.07	Average stream bank slope (°)	29.97	1.45	0.136	0.06
Indigenous vegetation (%)	37.57	1.85	0.044	0.07	Mosses/liverworts (%)	27.57	1.33	0.233	0.05
Scrub vegetation (%)	29.96	1.45	0.158	0.06	Shrubs (%)	27.54	1.33	0.196	0.05
Sandstone (%)	27.59	1.33	0.207	0.05	Vegetation ≤0.3 m in height (%)	27.09	1.31	0.241	0.05
Riparian buffer in woody vegetation (%)	25.39	1.22	0.248	0.05	Basal area (m^2 per 20 m^2)	25.24	1.21	0.252	0.05
Grass vegetation (%)	24.99	1.20	0.252	0.05	Vegetation >0.3 m in height (%)	23.84	1.14	0.315	0.05
Mudstone (%)	22.45	1.07	0.377	0.04	Cover bare ground (%)	17.13	0.81	0.655	0.03
Distance to native forest (m)	20.74	0.98	0.461	0.04					
Ashes older than Taupō eruption (%)	10.38	0.48	0.933	0.02					

Appendix 5: Physicochemical characteristics of 26 study sites in Pirongia, Whaingaroa and Whatawhata.

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Time collected	Temperature (°C)	DO mg/L	DO (% saturation)	Conductivity (µ/s @ 25°)	pH	Turbidity (NTU)	Hardness (as mg/L of $CaCO_3$)
MK1	PRG	Pasture	0	8.38	14.33	10.74	105.00	69.80	7.51	0.97	12
R1	PRG	Recent	2	8.35	12.85	11.05	110.00	71.80	7.73	1.57	16
TP3	PRG	Recent	2	8.37	12.05	10.41	96.70	58.30	7.50	1.83	6
KA1	PRG	Recent	3	9.26	16.90	9.27	95.90	110.0	8.13	2.44	47
TP5	PRG	Recent	4	8.49	13.74	10.23	97.90	68.60	8.00	1.26	8
TP6	PRG	Recent	4	9.31	14.29	10.64	103.90	72.10	7.56	3.24	18
MM1	PRG	Mid	6	8.43	14.52	10.55	103.70	57.50	7.60	1.76	10
TP2	PRG	Mid	8	9.41	11.10	10.64	96.80	110.80	7.73	2.57	22
O1	PRG	Mid	9	10.30	15.07	9.53	94.70	60.10	7.70	14.8	12
TP1	PRG	Mid-late	12	9.31	12.06	10.55	98.10	68.70	7.28	4.61	14
R2	PRG	Native	N/A	8.28	9.95	10.97	96.20	53.20	8.05	7.41	6
T1	WG	Pasture	0	9.17	13.94	10.46	101.40	79.80	6.62	6.38	20
BC2	WG	Recent	5	8.45	17.20	9.00	93.50	129.40	7.18	0.90	21
BC1	WG	Mid-late	11	8.28	16.40	9.89	101.11	95.10	7.54	4.09	23
TA1	WG	Mid-late	12	8.43	16.19	9.86	100.40	125.50	7.87	3.71	20
AA1	WG	Mid-late	13	9.03	15.12	9.99	99.40	106.8	7.85	3.45	14
OM1	WG	Mid-late	14	9.05	14.30	8.33	81.30	248.70	7.95	0.95	67
TM1	WG	Old	16	8.40	17.67	6.67	70.00	110.10	7.78	0.83	13
W1	WG	Old	23	8.31	14.92	9.80	100.10	132.90	7.34	1.68	22

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Time collected	Temperature (°C)	DO mg/L	DO (% saturation)	Conductivity (µ/s @ 25°)	pH	Turbidity (NTU)	Hardness (as mg/L of $CaCO_3$)
W2	WG	Native	N/A	8.31	11.68	11.10	103.3	113.80	8.30	2.77	20
K3	WW	Pasture	0	9.51	12.44	10.94	102.1	112.80	7.69	7.62	18
TU1	WW	Recent	1	8.15	15.60	9.41	94.5	83.10	8.58	9.52	27
TU2	WW	Recent	5	12.31	18.20	9.47	100.5	134.90	7.70	5.83	30
K1	WW	Old	18	8.13	14.40	9.70	95.3	84.80	7.50	8.64	13
K2	WW	Old	18	8.49	15.27	9.99	99.7	97.00	7.38	9.11	15
WK1	WW	Native	N/A	8.28	14.33	10.34	101.0	107.50	7.21	5.80	15

Appendix 6: Kōura characteristics of 26 study sites throughout Pirongia, Whaingaroa and Whatawhata.

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Total count	Mean length (OCL mm)	Minimum length (OCL mm)	Maximum length (OCL mm)	Median length (OCL mm)	Total female count	Total male count	Total unidentified count	Count mature females (≥17 mm OCL)	Count YOY (≤ 13 mm OCL mm)	Wetted area (m^2)	Number of kōura per m^2
MK1	PRG	Pasture	0	23	17.33	10	27.5	15.00	13	10	0	7	7	171.20	0.13
R1	PRG	Recent	2	63	13.41	7	30	12.00	28	31	4	5	42	224	0.28
TP3	PRG	Recent	2	88	10.55	2	22	10.00	26	26	36	10	64	38.70	2.27
KA1	PRG	Recent	3	3	9.33	6	16	6.00	1	0	2	0	2	103.50	0.03
TP5	PRG	Recent	4	12	15.04	11	24	15.00	7	5	0	0	4	38.10	0.31
TP6	PRG	Recent	4	130	13.31	2	27	12.75	46	73	11	14	78	191.00	0.68
MM1	PRG	Mid	6	76	15.82	3	23	16.00	33	41	2	18	18	155.30	0.49
TP2	PRG	Mid	8	91	9.73	2	24	10.00	22	27	42	3	69	57.10	1.59
O1	PRG	Mid	9	40	6.88	3	17	6.00	3	4	33	1	34	59.30	0.67
TP1	PRG	Mid-late	12	85	10.52	4	22	10.00	28	21	36	6	61	82.00	1.04
R2	PRG	Native	N/A	70	7.93	3	16	7.00	11	13	46	0	66	65.80	1.06
T1	WG	Pasture	0	21	15.43	11	23	14.00	9	10	2	4	7	80.10	0.26
BC2	WG	Recent	5	1	16.50	16.5	16.5	16.50	0	1	0	0	0	55.70	0.02
BC1	WG	Mid-late	11	16	12.66	6	21	12.00	7	8	1	0	10	69.80	0.23
TA1	WG	Mid-late	12	25	13.96	3	23	14.00	8	16	1	1	12	97.10	0.26
AA1	WG	Mid-late	13	36	11.88	2	22.5	11.50	15	18	3	2	26	101.00	0.36
OM1	WG	Mid-late	14	0	0.00	0	0	0	0	0	0	0	0	62.40	0.00
TM1	WG	Old	16	1	19.50	19.5	19.5	19.50	0	1	0	1	0	85.00	0.01
W1	WG	Old	23	41	14.59	8	25	14.00	18	23	0	7	17	138.60	0.30
W2	WG	Native	N/A	21	11.69	6	20	11.50	12	6	3	3	17	165.50	0.13

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Total count	Mean length (OCL mm)	Minimum length (OCL mm)	Maximum length (OCL mm)	Median length (OCL mm)	Total female count	Total male count	Total unidentified count	Count mature females (≥17 mm OCL)	Count YOY (≤ 13 mm OCL mm)	Wetted area (m^2)	Number of kōura per m^2
K3	WW	Pasture	0	57	13.93	7.5	23	13.00	25	30	2	6	30	104.70	0.54
TU1	WW	Recent	1	5	13.80	5	22.5	14.00	2	1	2	2	2	76.20	0.07
TU2	WW	Recent	5	9	10.17	4.5	25	6.00	2	1	6	1	6	113.90	0.08
K1	WW	Old	18	173	12.46	3	34	11.00	73	81	19	16	128	68.10	2.54
K2	WW	Old	18	26	11.60	3	23	11.00	12	11	3	0	20	113.50	0.23
WK1	WW	Native	N/A	30	10.57	4	21.5	10.25	8	12	10	1	21	181.80	0.17

Appendix 7: Marginal tests for DistLM analysis for upstream catchment, instream habitat, riparian and physicochemical variables for total kōura density dissimilarity matrix for 26 study sites throughout Pirongia, Whaingaroa and Whatawhata.

Upstream catchment					Instream habitat				
Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.	Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.
Catchment area (ha)	8689.30	5.55	0.003	0.19	Macrophyte cover (%)	12589	8.98	0.001	0.27
Scrub vegetation (%)	7129.30	4.37	0.023	0.15	Run habitat (%)	6789.70	4.13	0.015	0.15
Change in gradient (m/m)	6437.30	3.88	0.021	0.14	Root complexes (m^2)	6397.50	3.85	0.012	0.14
“Other” vegetation (%)	4160.80	2.37	0.074	0.09	Gravel substrate (%)	5359.40	3.15	0.034	0.12
Distance to native forest (m)	3249.70	1.81	0.143	0.07	Overhanging bank vegetation (%)	5046.10	2.94	0.038	0.11
Volcanic rock (%)	2018.30	1.10	0.32	0.04	Gradient change over reach (m/m)	4555.40	2.62	0.045	0.10
Riparian buffer in woody vegetation (%)	2011.60	1.09	0.332	0.04	Channel width (m)	4340.90	2.49	0.079	0.09
Indigenous vegetation (%)	1445.60	0.77	0.495	0.03	Cobble substrate (%)	3440.70	1.93	0.125	0.07
Ashes older than Taupō eruption (%)	1411.60	0.76	0.505	0.03	Shade (% canopy cover)	3299.60	1.84	0.149	0.07
Grass vegetation (%)	1016.30	0.54	0.671	0.02	Riffle habitat (%)	3240.70	1.81	0.134	0.07
					Silt substrate (%)	2986.50	1.66	0.165	0.06
					“Other” habitat (%)	2006.30	1.09	0.31	0.04
					Thalweg depth (m)	1931.60	1.05	0.34	0.04
					Pool habitat (%)	1728.80	0.93	0.439	0.04
					Boulder substrate (%)	1640.30	0.88	0.459	0.04
					Total undercuts (m)	899.42	0.48	0.718	0.02
					Sand substrate (%)	869.51	0.46	0.712	0.02
					Clay substrate (%)	688.22	0.36	0.669	0.01
					Total wood (m^2)	441.49	0.23	0.901	0.01
					Bedrock substrate (%)	431.14	0.23	0.99	0.01

Riparian Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.	Physicochemical Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.
Average stream bank slope (°)	3627.50	2.04	0.110	0.078	Dissolved oxygen (mg/L)	12395	8.79	0.001	0.27
Basal area (m^2 per 20 m^2)	3111.10	1.73	0.158	0.067	Temperature (°C)	10171	6.77	0.001	0.22
Shrubs (%)	2472.50	1.36	0.240	0.053	Hardness (as mg/L of $CaCO_3$)	8354	5.29	0.003	0.18
Leaf litter (%)	2418.40	1.32	0.242	0.052	Conductivity (µS/cm)	7946.50	4.98	0.004	0.17
Vegetation ≤0.3 m in height (%)	1655.50	0.89	0.426	0.036	Turbidity (NTU)	3570.20	2.01	0.111	0.08
Mosses/liverworts (%)	845.22	0.45	0.693	0.018	pH	1456.70	0.78	0.497	0.03
Vegetation >0.3 m in height (%)	757.65	0.40	0.794	0.016					
Number of saplings (per m^2)	563.38	0.30	0.882	0.012					
Average buffer width (m)	548.83	0.29	0.860	0.012					
Bare ground (%)	503.99	0.26	0.921	0.011					

Appendix 8: Variables included in overall model for kōura density

Instream habitat	Upstream Catchment	Riparian	Physicochemical
Run habitat (%)	Catchment area (ha)	Average stream bank slope (°)	Dissolved oxygen (mg/L)
Macrophyte cover (%)	“Other” vegetation (%)	Number of saplings (per m^2)	Hardness (as mg/L of $CaCO_3$)
Root complexes (m^2)	Ashes older than Taupō eruption (%)		
Gravel substrate (%)			
Channel width (m)			
Cobble substrate (%)			
Silt substrate (%)			
Boulder substrate (%)			
Sand substrate (%)			
9 variables	3 variables	2 variables	2 variables

Appendix 9: Marginal tests for DistLM analysis for instream habitat variables for YOY kōura density dissimilarity matrix for 26 study sites throughout Pirongia, Whaingaroa and Whatawhata.

Variable	SS(trace)	Pseudo-F	P value	Proportion R^2
Macrophyte cover (%)	7828.50	5.88	0.002	0.22
Root complexes (m^2)	7726.60	5.78	0.009	0.22
Gravel substrate (%)	6872.80	4.99	0.009	0.19
Sample reach gradient (m/m)	6716.10	4.85	0.015	0.19
Channel width (m)	48910	3.32	0.050	0.14
Thalweg depth (m)	3199.60	2.06	0.115	0.09
Shade (% canopy cover)	3002.20	1.92	0.138	0.08
Run habitat (%)	2642.60	1.67	0.209	0.07
Other habitat (%)	2607.40	1.65	0.166	0.07
Boulder substrate (%)	2463.70	1.55	0.223	0.07
Cobble substrate (%)	2162.90	1.35	0.259	0.06
Length of undercuts (m)	1983.60	1.23	0.277	0.06
Silt substrate (%)	1397.70	0.85	0.447	0.04
Pool habitat (%)	1251.30	0.76	0.476	0.03
Sand substrate (%)	1249.20	0.76	0.465	0.03
Riffle habitat (%)	1229.50	0.75	0.503	0.03
Overhanging bank vegetation (%)	828.01	0.50	0.629	0.02
Wood (m^2)	560.31	0.33	0.909	0.02
Bedrock substrate (%)	505.34	0.30	0.836	0.01
Clay substrate (%)	192.47	0.11	0.953	0.01

Appendix 10: Fish community characteristics for 26 study sites throughout Pirongia, Whaingaroa and Whatawhata.

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Number of eels captured (count)	Eel average length (mm)	Eel minimum length (mm)	Eel maximum length (mm)	Eel median length (mm)	Number of eels captured ≤ 300mm	Number of eels captured ≥301 mm	Number of bullies captured (count)	Number of galaxiids captured (count)	Number of trout captured (count)	Number of other fish captured (count)	Distance to sea (km)
MK1	PRG	Pasture	0	7	465.14	120	1000	490	2	5	91	1	4	1	157.4
R1	PRG	Recent	2	8	273.12	120	490	234	5	3	13	0	26	1	152.9
TP3	PRG	Recent	2	2	175	100	250	175	2	0	0	0	0	0	152.9
KA1	PRG	Recent	3	11	236.45	89	530	200	8	3	50	0	1	0	148.5
TP5	PRG	Recent	4	2	535	60	1010	535	1	1	0	0	0	0	150.5
TP6	PRG	Recent	4	28	231.25	60	630	182.5	19	9	26	0	0	0	150.6
MM1	PRG	Mid	6	4	198	173	214	202.5	4	0	0	0	0	0	155.7
TP2	PRG	Mid	8	2	530	420	640	530	0	2	0	0	0	0	152.5
O1	PRG	Mid	9	15	335	80	710	320	7	8	0	0	0	0	182.9
TP1	PRG	Mid-late	12	4	548.75	300	830	532.5	1	3	0	0	0	0	153.8
R2	PRG	Native	N/A	1	110	110	110	110	1	0	0	0	0	0	154.5
T1	WG	Pasture	0	57	211.77	80	900	140	46	11	0	0	0	0	12.97
BC2	WG	Recent	5	45	212.66	60	620	130	35	10	0	53	0	0	1.6
BC1	WG	Mid-late	11	4	287.50	60	820	135	3	1	2	15	0	0	1.9
TA1	WG	Mid-late	12	51	207.64	75	680	160	40	11	5	1	0	0	4.3
AA1	WG	Mid-late	13	50	123.20	50	410	90	46	4	8	0	0	0	7.3
OM1	WG	Mid-late	14	77	87.66	0	405	70	76	1	45	126	0	0	1.1
TM1	WG	Old	16	27	195.55	85	564	137	23	4	0	1	0	0	2.7
W1	WG	Old	23	40	107.62	40	340	80	37	3	37	3	0	0	2.0
W2	WG	Native	N/A	23	203.47	50	525	150	17	6	30	1	0	0	8.6

Site	Spatial cluster	Riparian planting age group	Riparian planting age	Number of eels captured (count)	Eel average length (mm)	Eel minimum length (mm)	Eel maximum length (mm)	Eel median length (mm)	Number of eels captured ≤ 300mm	Number of eels captured ≥301 mm	Number of bullies captured (count)	Number of galaxiids captured (count)	Number of trout captured (count)	Number of other fish captured (count)	Distance to sea (km)
K3	WW	Pasture	0	6	195.33	130	320	165	5	1	123	0	0	0	122.0
TU1	WW	Recent	1	47	154.10	80	790	110	44	3	224	2	0	0	128.3
TU2	WW	Recent	5	29	135.17	70	355	115	28	1	48	6	0	0	125.3
K1	WW	Old	18	10	112.50	90	150	110	10	0	0	0	0	0	122.0
K2	WW	Old	18	19	173.15	50	620	80	15	4	79	0	0	0	122.4
WK1	WW	Native	N/A	3	223.33	60	540	70	2	1	7	2	0	0	122.0

Appendix 11: Marginal tests for DistLM analysis for upstream catchment, instream habitat and riparian variables for eel density dissimilarity matrix for 26 study sites throughout Pirongia, Whaingaroa and Whatawhata.

Upstream catchment					Instream habitat				
Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.	Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.
Distance downstream to the sea (km)	11508	8.68	0.010	0.26	Pool habitat (%)	218.39	0.12	0.926	0.01
Grass vegetation (%)	7802.10	5.27	0.012	0.18	Bedrock substrate (%)	790.47	0.45	0.679	0.02
Indigenous vegetation (%)	7623.80	5.13	0.017	0.18	Sand substrate (%)	766.09	0.43	0.658	0.02
Gradient (m/m)	6554	4.28	0.025	0.15	Gravel substrate (%)	826.59	0.47	0.633	0.02
“Other” vegetation (%)	2571.90	1.51	0.235	0.06	Clay substrate (%)	1046.7	0.59	0.539	0.02
Mudstone (%)	1654.10	0.95	0.431	0.04	Thalweg depth (m)	1251.5	0.71	0.486	0.03
Catchment area (ha)	1495.10	0.86	0.410	0.03	Silt substrate (%)	1178.1	0.67	0.485	0.03
Sandstone (%)	1173.70	0.67	0.548	0.03	Cobble substrate (%)	1221.9	0.70	0.455	0.03
Scrub vegetation (%)	956.30	0.54	0.629	0.02	Length of undercuts (m)	1460.8	0.84	0.401	0.03
Volcanic rock (%)	733.90	0.41	0.645	0.02	Wood (m^2)	2071.9	1.21	0.283	0.05
Ashes older than Taupō eruption (%)	706.90	0.40	0.689	0.02	Boulder substrate (%)	2164.8	1.26	0.258	0.05
Riparian buffer in woody vegetation (%)	685.20	0.39	0.700	0.02	Sample reach gradient (m/m)	2339.4	1.37	0.253	0.05
					Riffle habitat (%)	2489.8	1.46	0.232	0.06
					Shade (% canopy cover)	2665	1.57	0.217	0.06
					Channel width (m)	2531.3	1.49	0.214	0.06
					“Other” habitat (%)	3648.8	2.21	0.129	0.08
					Root complexes (m^2)	3994.1	2.44	0.103	0.09
					Overhanging bank vegetation (%)	3670.7	2.22	0.096	0.08
					Run habitat (%)	4591.3	2.85	0.069	0.11
					Macrophyte cover (%)	9611.5	6.84	0.002	0.22

Riparian

Variable	SS(trace)	Pseudo-F	P value	Proportion R^2.
Average stream bank slope (°)	4863.90	3.04	0.063	0.11
Average buffer width (m)	2951	1.75	0.171	0.07
Bare ground (%)	2373.10	1.39	0.243	0.05
Number of saplings (per m^2)	2275.10	1.33	0.266	0.05
Mosses/liverworts (%)	823.34	0.46	0.619	0.02
Leaf litter (%)	654.45	0.37	0.678	0.02
Vegetation >0.3 m (%)	465.51	0.26	0.824	0.01
Basal area (m^2 per 20 m^2)	379.19	0.21	0.844	0.01
Shrubs (%)	203.74	0.11	0.921	0.00
Vegetation ≤0.3 m (%)	176.15	0.10	0.953	0.00