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The Effects of Stream Riparian Cover and Insect Contributions to the Diet of Banded Kokopu (Galaxias fasciatus)

A thesis submitted in partial fulfilment of the requirements for the degree of

Master of Science in Environmental Science

By

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ABSTRACT

Banded kokopu (*Galaxias fasciatus*) are a native endemic fish that inhabit many first to third order streams and rivers throughout lowland New Zealand. Their presence is reliant on stream shading, cool water temperature, riparian vegetation cover, adequate habitat and availability of invertebrates for consumption, particularly invertebrates of terrestrial origin. This study tested whether the extent of riparian cover in forested streams affected terrestrial insect contributions to the stream. This study also tested whether varied insect availability in drift then influenced banded kokopu diet.

Eight forest sites were chosen throughout the Waitakere Ranges to represent a range of vegetation cover types, from 65-100% overhead vegetation cover. All sites were assessed during summer, and four of those eight sites during winter, to provide seasonal comparison while maintaining a cross section of the cover variable. Measurements of allochthonous invertebrate drift were obtained during both day and night. Fish gut contents were also obtained from each site.

This study found that significantly higher densities of insects in drift occurred in summer during both day and night time collections. Although many significant relationships were found between cover and insect abundance during day and night at summer and winter, there was no unequivocal evidence to confirm that increases in riparian cover above 65% caused a subsequent increase in the density of terrestrial insects in drift.

Banded kokopu diet consisted of 87.6% terrestrial components during summer and 58.7% during winter, giving an average diet composition of 73.1% terrestrial insects. There was little evidence to suggest that banded kokopu diet tracked the provision of terrestrial prey abundances in drift. Banded kokopu were selective in their consumption habits, consuming a similar community of insects from the proportion of those available.

Comparison of these results with other studies suggest that abundant food was available to fish in streams utilised in this study, allowing banded kokopu to be selective in their diet choices from the proportion of food opportunities available. However, a lack of suitable feeding habitat and residential habitat restricted the abundance of banded kokopu at these locations. It is possible that where dense populations of banded kokopu exist, increased competition with conspecifics may cause prey consumption to closer track diet availability.
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“If I have seen a little further it is by standing on the shoulders of Giants”

Sir Isaac Newton - 1676
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Chapter 1 | P.G. Hancock

1

GENERAL INTRODUCTION

1.1 NEW ZEALAND FRESHWATER ISSUES AND IMPACTS

Since New Zealand became inhabited about 700-800 years ago, our country has undergone significant deforestation from about 85% forested land cover to approximately 23% at present (Statistics New Zealand, 2008). The Waitakere Ranges was one such area, where heavy deforestation occurred from 1840’s - 1930’s to harvest trees for wood and clear land for habitation (Harvey and Harvey, 2006). A nationwide reduction in native forest cover has significantly altered freshwater ecosystems that were once surrounded by this native forest. This historic deforestation has left few areas of lowland forest in New Zealand, resulting in a loss of viable habitat for many fish species (McDowall, 2006).

Increased pressure placed by humans on freshwater resources and a continued decrease in native freshwater biodiversity can produce undesirable consequences for receiving waterways in New Zealand (Allan and Castillo, 2007), particularly to native stream fishes (McDowall, 2006). A high percentage of the land area is utilised for agricultural purposes, and this land use can reduce fish habitat, leach excess nutrients, sediments and biological contaminants into waterways (Harding et al., 2004; Cullen et al., 2006). Exotic forestry can cause large fluctuation effects on water yield, stream cover, water temperature, nutrients and suspended sediment from before and after harvesting periods (Boothroyd et al., 2004; Harding et al., 2004; Quinn et al., 2004). Water impoundment and water abstractions can alter water quality, reduce flow rates and cause fish migration barriers (Harding et al., 2004). Although spatially localised in New Zealand, populated urban areas and their associated impervious surfaces can have detrimental effects on streams (Surrey, 2004). This is particularly relevant around the cities where there is loss of habitat, polluted runoff, high peak flow rates, fish migration barriers and loss of riparian vegetation (Harding et al., 2004; Roy et al., 2007). Introduced species of aquatic flora and fauna also threaten to out-compete native species by restricting the amount and quality of viable habitat for native species (McDowall, 2006).

1 The Waitakere Ranges has since been reverted back to native forest, currently the Waitakere Ranges Regional Park managed by Auckland Regional Council.

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All of these above influences have the ability to significantly alter channel morphology, habitat, flow regimes, water quality and consequently, stream ecology. A loss of biodiversity through this loss of effective habitat for native species that live in these streams can cause them to become restricted in abundance across the country. An ongoing challenge facing land managers is how to best avoid, remedy or mitigate these respective impacts on the environment in a practical and effective manner. As our knowledge of environmental impacts increase, so too must our ability to provide practical and viable solutions to sustain the core values that are placed on these receiving environments.

### 1.2 Restoration and Rehabilitation

When improving an already degraded stream environment, Bradshaw (1996) and Harper et al. (1998) have defined two distinct management perspectives, being either ‘restoration’ or ‘rehabilitation’ (Harding et al., 2004). Restoration implies returning the environment to a near original state and rehabilitation implies an improvement to a less damaged state. It is important to recognise that these two goal definitions are used interchangeably, but have functionally different descriptions.

Based on our current values and current knowledge of ecosystems, humans employ a number of initiatives to counteract the impacts caused by land use activities on the environment. Two methods employed to reduce/eliminate environmental impacts are either ‘active’ and/or ‘passive’ methods of treatment. Active methods of environmental protection tend to be expensive to construct and maintain. These methods are commonly used in commercial activities and their application is commonly used to treat point source pollution, before releasing it to the environment in a less toxic form. An example of this is the treatment of raw human sewerage in a waste water treatment plant, before finally releasing it into the environment. Passive treatment methods tend to be relatively inexpensive to construct and maintain when compared to active treatment systems and can be implemented across a larger spatial scale. These methods are favoured for impacts to the environment that occur over a large area. An example of this is the provision and planting of vegetation alongside freshwater streams to reduce diffuse impacts from land uses such as agriculture (Harding et al., 2004; Cullen et al., 2006) or forestry (Boothroyd et al., 2004).

The implementation of restoration initiatives to increase water quality is dependent on the scale of the problem, i.e. it can range from catchment-wide changes that affect water quality, to small microhabitat effects that only affect a localised area of stream. The larger the spatial area, the more complex and difficult a restoration activity will be to implement (Figure 1-1). Chronic water quality degradation that occurs from cumulative effects over a wide area can be a large-scale problem and will require catchment wide planning to negate these effects. Alternatively, small-scale problems at
one site may only require local solutions. It is also important to consider the structure of riparian vegetation in natural forest systems change along a continuum from mountains to sea. This altitudinal gradient leads to changes in environmental parameters such as temperature and landscape processes also predictably change from mountains to sea (Harding et al., 2004). The consideration of scale is important when considering management towards achieving environmental improvement goals.

**Figure 1-1** Conceptual diagram showing the importance of scale for both restoration targets and activities (Figure 38.3 from Harding et al., 2004)

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### 1.3 RIPARIAN ZONES FOR STREAM HEALTH

The word ‘riparian’ comes from Latin *ripa*, meaning river bank (Agnes, 2005). The riparian zone is a corridor of land immediately adjoining either side of the stream. It has importance for bank stability, biodiversity and significantly influences the ecology outside and inside of the stream. It is commonly utilised as a passive method for the treatment of land that improves both stream water and habitat quality (Broadmeadow and Nisbet, 2004).

Riparian zones are being encouraged and utilised as mitigation measures against a range of human land use effects on freshwater stream environments in New Zealand (Hickey and Doran, 2004). They can act to filter sediments from surface runoff, take up excess nutrients from shallow subsurface water flows, provide shade to mediate and maintain cool water temperatures, and stabilise stream banks (Rowe et al., 2002; Boothroyd et al., 2004; Hickey and Doran, 2004; Niyogi et al., 2007).

Vegetated riparian zones improve water quality by acting as bio-filters to intercept surface and shallow sub-surface flow, before drainage water from surrounding land reaches the stream, as illustrated in Figure 1-2. Sediment is removed from water by filtration, pollutants bind to soil and nutrients are taken up by plants before reaching the stream (Harding et al., 2004). If these pollutants are contributed to the stream in excessive amounts, they can become damaging to the
stream ecosystem and to human health (Rowe et al., 2000; Niyogi et al., 2007). The root systems of riparian vegetation bind stream banks together, serving the dual purpose of stabilising the substrate to reduce erosion damage and reducing the amount of suspended sediment eroded from the banks during high flows. The role of riparian zones in reducing land use effects is determined by the size, age and species of vegetation (Naiman and Decamps, 1997) and the extent of inflow that bypasses riparian filtration (i.e., storm water drainage pipes from impervious surfaces). The ability for riparian zones to mitigate the effects of some intensively utilised land uses may be limited, particularly in urban areas (EnviroVentures and Associates Ltd. et al., 2004; Roy et al., 2007). The riparian zone contributes proportionately less influence on the stream as stream width increases (Naiman and Decamps, 1997).

Riparian zones provide shading for streams, mediating stream temperatures and keeping them cool during the daytime when direct solar radiation would otherwise warm the stream (Figure 1-3). Cool water maintains high levels of dissolved oxygen, which is important for the survival of many aquatic organisms. Riparian vegetation can also maintain a humid microclimate close to the stream, which is important for some insect life stages and may be important for egg hatching of some native Galaxiid fish species (Charteris et al., 2003).

Riparian zones contribute energy to the stream in the form of leaves, twigs, woody debris and insects (Figure 1-3). These are referred to as energy subsidies, donations or allochthonous inputs.
(Chan et al., 2008), all being organic inputs which have originated from the terrestrial zone. Leaves, twigs and wood combine during high flow events to form discrete aggregations, termed ‘debris dams’ (Figure 2-13). These dams provide important functional habitat for macroinvertebrates that live on the debris and also for native fish such as Anguilla eel species and banded kokopu (Galaxias fasciatus; McCullough, 1998) that seek cover amongst the debris. Riparian vegetation organic matter contributed to streams provides an important source of nourishment for primary consumer aquatic organisms.

Riparian zones are valued for maintaining biodiversity in the terrestrial landscape by providing wildlife habitat (Naiman and Decamps, 1997). They are often termed ‘corridors’ for dynamic migration and dispersal of terrestrial floral and faunal species between patches of isolated forest. This increases connectivity of the landscape and encourages species migration, species diversity and genetic diversity. A component of the diversity present within vegetated zones is that of insect activity. Vegetated riparian zones provide essential habitat and food for a diverse range of insects that live within it and are reliant on it (Collier and Smith, 2000). A study undertaken by Huryn and Edwards (1996) showed terrestrial invertebrate contributions to streams surrounded by tussock or native riparian vegetation were significantly higher than inputs from pasture. They illustrate that riparian land use can be an important determinant of the availability of terrestrial insects to stream fishes. Communities of native fish in streams can be enhanced where vegetated strips are present along a riparian margin (Rowe et al., 2002).

Many overseas studies have highlighted that leaf inputs of deciduous vegetation during autumn are important in controlling stream food webs (Mason and Macdonald, 1982; Dineen et al., 2007). This causes great fluctuation in food web structure with widely variable amounts of allochthonous inputs each season. New Zealand biogeography is unique in that native forests consist largely of evergreen
vegetation that does not provide highly varied annual leaf fall inputs to streams. This limits any direct comparisons with overseas studies that assess riparian vegetation and the contribution of terrestrially derived invertebrates to the stream when their vegetation is primarily deciduous. Overseas studies of insect inputs from these systems have shown that on average, deciduous woodland produces more biomass of terrestrial insects than coniferous vegetation (Allan et al., 2003).

The abundance of terrestrial invertebrates in stream drift showed seasonal fluctuation in studies under deciduous vegetation (Romero et al., 2005) and in pasture streams (Edwards and Huryn, 1995). A review of the current literature could find little information on the seasonal fluctuation of allochthonous insect inputs under primarily evergreen vegetation. This study will shed some light on the fluctuation of allochthonous drift activity between summer and winter seasons under native New Zealand evergreen vegetation

### 1.4 BANDED KOKOPU

Invertebrates that originate from the terrestrial environment provide an important energy source for many freshwater fish species (McDowall et al., 1996; Bonnett and Lambert, 2002; West et al., 2005; Chan et al., 2008). One such native fish that primarily feeds on terrestrial invertebrates is the banded kokopu (West et al., 2005), a member of the Galaxiidae family. Banded kokopu exhibit a diel activity regime that is primarily nocturnal, but shows an activity increase around sunrise and sunset (crepuscular activity; McCullough and Hicks, 2002). They utilise a specialised olfactory and lateral line sensory system that enables them to detect and locate prey in dark light environments (Halstead, 1994; Baker, 2000). The immature form of banded kokopu is one of five species that comprise the commercial and recreational whitebait catch and the species is valued as tāonga by Māori (DOC, 2005).
Figure 1-4  Figures a)left, b)top right and c) bottom right; show banded kokopu (*Galaxias fasciatus*). Figure b) and c) show the thin vertical bands running vertically on the caudle peduncle, below the adipose fin. These lines are defining characters useful for identification. Note the darker colouration and mottled colour on top of head in Figure c).

Banded kokopu exhibit a cryptic colouration that is noticeably varied within and between streams (Figure 1-4). They are identified by thin vertical bands that run up their sides to their back. Banded kokopu are commonly found in first to third order streams of under two meters width (Rowe et al, 2002) that are close to the coast in lowland New Zealand. They are noticeably absent around the mainly agricultural lowland areas of the Canterbury Plains and Manawatu areas (Figure 1-5). They prefer pools and backwaters of slow moving water (~0.05m/s⁻¹) and optimum depths of around 0.8m (McCullough, 1998).
Figure 1-5  Map of New Zealand and the Chatham Islands showing distributions of *Galaxias fasciatus* from New Zealand Freshwater Fish Database records (NIWA, 2008) from 1923-2008.

Banded kokopu prefer high proportions of riparian shading in their habitat (Baker and Smith, 2007) and are less abundant or even absent from streams that do not have stream shading, such as pasture sites (Rowe et al., 1999). Their preference for areas with high riparian cover could be a component linked to the high percentage of terrestrially derived invertebrates found in their diet. It is possible that a proportion of terrestrial invertebrates are blown or dislodged from overhanging vegetation. As mentioned in Section 1.2, riparian cover over the stream is the primary determinant mediating cool stream temperatures. In a controlled laboratory experiment undertaken by Richardson, Boubee et al. (1994) banded kokopu preferred water temperatures between 16.1-17.3°C, which is relatively cooler than temperature preferences of other native freshwater fish.

Banded kokopu commonly reside in bush covered streams with undercut banks, leaf, twig and woody debris dams for in-stream cover (Hopkins, 1979; Baker and Smith, 2007). The fish exhibit a preference to utilise undercut banks with exposed root matter as cover, as well as debris dams and
coarse boulder cover (McCullough, 1998; Rowe and Smith, 2003; Baker and Smith, 2007). Overhead canopy cover and terrestrial twigs and leaves alongside the stream also provide significant spawning habitats by moderating climate and reducing frost, maintaining high relative humidity and excluding stock from the stream to protect eggs from being trampled (Charteris et al., 2003). Banded kokopu habitat can become eliminated or degraded where there are spawning barriers, deforestation, land use intensification, introduced species that displace natives, water impoundment and water abstraction (McDowall, 2006).

1.5 TERRESTRIAL INVERTEBRATE CONTRIBUTIONS TO THE STREAM

Previous research has highlighted the importance of riparian vegetation as a source of arthropod inputs to stream systems (Edwards and Huryn, 1996; Chan et al., 2008). Much of this research has focused on influences to diets of salmonid fishes (Edwards and Huryn, 1995; Nakano et al., 1999a; Kawaguchi and Nakano, 2001; Allan et al., 2003; Romero et al., 2005). As mentioned in Section 1.3, a large portion of overseas research has investigated invertebrate contributions from deciduous vegetation, which is functionally different to that from evergreen vegetation when considering allochthonous contributions to streams. The present research will aim to fill knowledge gaps surrounding temporal variation (day Vs night; summer Vs winter) in the provision of allochthonous invertebrate inputs from native evergreen riparian vegetation to stream drift.

Prior studies of stream cover have looked at the relationship between stream cover and periphyton growth (Boothroyd et al., 2004), cover and macroinvertebrate assemblages (Quinn et al., 2004), and cover and terrestrially derived invertebrate contributions (Edwards and Huryn, 1996). The latter study looked at the differences between stream cover in vastly contrasting land uses when considering allochthonous invertebrate inputs. Little research has been undertaken to further characterise the nature of this riparian cover over streams and relate this to allochthonous invertebrate inputs. This current research aims to achieve this by assessing the invertebrate contribution to stream drift and its relationship with vegetative cover extent, and more specifically, the characterisation of vegetation cover proximity to the stream in native forests.

Data on the diet of banded kokopu has been previously studied (Main and Lyon, 1988; West et al., 2005), both of these studies concluded that terrestrial insects comprise the majority diet of banded kokopu. The study by Main and Lyon (1988) also assessed the composition and abundance of aquatic and terrestrial drift, to provide a comparison between species observed in drift and species observed in the banded kokopu gut. The present study will utilise similar data collection and assessment, but additionally will aim to correlate insect abundance to stream cover and the
proximity of this cover to the stream surface. Rowe et al. (1999) allude to understory vegetation such as native shrubs and ferns possibly playing a more important role than the overstory forest type (exotic or native forest) when considering fish habitat. The present study aims to quantify understory and overstory cover proximity of vegetation and its role in determining invertebrate contributions to the stream.

1.6 OBJECTIVES OF STUDY

Terrestrially derived food sources comprise the majority of banded kokopu diet, but how varied forest vegetation canopy cover influences terrestrially derived invertebrate inputs to banded kokopu diet has not been investigated. This project aims to quantify the contribution of terrestrially derived insects from stream riparian vegetation cover, and assess how these contributions affect the diet of banded kokopu (*Galaxias fasciatus*) in the Waitakere Ranges.

To assess this aim, the research is broken down into two main objectives:

- Determine how the extent of forest riparian vegetation cover influences the influx of terrestrially derived invertebrate contributions to the stream system.
- Assess the actual diet of banded kokopu at these locations to determine if feeding preferences track diet availability.

This study will assess whether there is a contributory relationship linking the extent of riparian canopy and riparian tier structure with fish diet, through the availability and subsequent consumption of terrestrial invertebrates.

To test these research objectives, two hypotheses have been constructed:

- Increased riparian vegetation canopy cover will increase the amount of terrestrially derived invertebrate inputs to the stream system.
- The proportion of terrestrially derived invertebrate food in the diet of banded kokopu will increase where riparian cover is greatest.
1.7 EXPERIMENTAL DESIGN

The study utilised a hierarchical nested design that allows the user to interrogate data on a number of levels. It utilises sub-sampling at the lowest hierarchical level to increase the precision from which we can estimate the response for each replicate (Gotelli and Ellison, 2004). Sites are nested within Seasons, Day/Night periods within Sites and Sampler Replicates are nested within each Day/Night period. This layout allows the user to interrogate data for differences between Seasons, Day/Night, and between Sampler Replicates (sub-samples; Figure 1-6 and Figure 1-7). It also allows the user to test for differences among replicates within treatments (Gotelli and Ellison, 2004) and to quantify the amount of error between measurements at each level in the hierarchy.

Figure 1-6 illustrates the nested sample program design layout during the summer sampling period for eight sites and Figure 1-7 illustrates the same for the winter sampling period, which only consists of four sites. Each Site will be sampled three times during the night and three times during the day. During each Day or Night sampling period, there will be three sub-samplers collecting drift from along the same stream reach. These replicates should increase the power of an estimate of the mean for each Day or Night treatment.
Utilising a nested design enables variation in the data to be assessed at each hierarchical level, explaining which layer determines the majority of the variation in the dataset. Figure 1-8 details the suite of measurements that were taken at the 'Site' and 'Sampler Replicates' level in the hierarchy. The parameter mean can be determined and analysed at each of the hierarchical levels above i.e., drift insect species can be assessed for error between replicates, between day or night periods, between sites or between seasons.
Figure 1-8 Diagram shows the parameters measured at each hierarchical level.
SITE SELECTION AND STUDY METHODS

2.1 SITE SELECTION

Eight different stream reaches were chosen according to the selection criteria detailed below. Sampling at all of these eight sites took place late January to early March, representing a 'summer' sampling period. A second sampling period took place mid-August to mid-September; this was termed a 'winter' sampling period. Note must be taken that these definitions of 'summer' and 'winter' have been loosely applied to each sampling period. They are intended to provide a seasonal comparison as it would be unrealistic to sample throughout the whole summer and the whole winter season given the scope and timing of this project.

The presence of banded kokopu is strongly associated with the presence of riparian vegetation (McDowall, 1990; Rowe et al., 1999). For sites to have banded kokopu present, it was assumed that 50-100% stream cover was required. This study aimed to assess the variation in stream cover within this 50-100% cover range. However, there was some difficulty in selecting sites at the lower end of this cover spectrum from around the Waitakere Ranges. This was because streams with low percentage cover were commonly coupled with other land use effects associated with lower catchment sites, particularly within agricultural, peri-urban and urban areas. This had the potential to introduce large errors where other land use variables were associated with the riparian cover variable, potentially making it difficult to distinguish between the effects of stream cover and the effects of catchment land use. The opposite occurred when assessing streams with native forested catchments, where most stream sites had a high degree of riparian cover due to well established vegetation growth.

The winter sampling period was reduced from eight original summer sites, to four of the original eight sites during winter. This was done for a number of reasons:

- The four sites selected for the winter sampling round recorded a high number of banded kokopu and still represented a good cross section of vegetation cover parameters. Therefore, the loss of four sites would reduce a large portion of time and cost, with a small proportionate loss of project detail. This greatly improved the efficiency of the project and allowed more time for data analysis.
• Although banded kokopu were present at all sites, there was a low number captured at four of the sites, which would limit the statistical rigidity of fish data obtained from those sites.
• The project design required a large quantity of insect identification. This proved to be a time consuming task. Given the timeframe, it was not realistic to undertake the same amount of effort for a winter sampling round.

The eight original study sites were chosen by way of a selection filter. This filter is a descriptive set of attributes that each site must possess in order for it to be suitable for the purposes of this study. A further set of site attributes gave the site a selection advantage over others, but were not essential.

**Region/Area:** All sites must exist within the vicinity of the Waitakere Ranges in the Auckland region.

**Access:** To accommodate for regular visits and the transport of heavy sampling equipment, an immediately accessible roadside site was preferable. A 15 minute walk from the roadside was considered the maximum practical distance to commute to a site.

**Contamination:** To eliminate any effects that the Auckland City water catchment area dams may have had on fish migration, no sites were located inside the Waitakere Ranges water catchment area. Native forest catchments with less than 10% developed land area were selected, to eliminate any effect human activities may have had on water quality or stream dynamics. Riparian vegetation must be wholly native, with little exotic vegetation present.

**Stream Characteristics:** Stony bottomed streams were chosen. This was to reduce the effect that varied substrate composition may have had on macroinvertebrate communities. It was preferable that the stream had areas of shallow (~0.8m) slow moving (~0.05m/s⁻¹) water as this is preferred as optimum depth and water speed for banded kokopu (McCullough, 1998). Another important physical attribute for banded kokopu resident habitat was the requirement for dense vegetative cover over the stream (Baker and Smith, 2007). Vegetative cover works to provide and maintain important physical and chemical attributes of the stream system, therefore streams had to be surrounded by native riparian vegetation that contributed leaves, twigs and branches. These components then combine to form woody debris dams that occasionally accumulate along a stream reach, creating favourable residential habitat for banded kokopu.

Undercut banks with tree roots and tree root mats are also considered favourable habitat. The presence of debris dams and undercut banks at sites was not considered essential, but were advantageous contributions to ideal habitat. A lack of vegetation draped in the stream was also an important prerequisite for study sites, enabling the collection of surface drift from a length of stream with an undisturbed water surface.
Distance from the ocean along the stream path was also considered during site selection, with distances between 1km and 20km from the ocean being preferable. This ensured that long migratory distances outside this range did not retard the numbers of fish available at each site.

Presence of banded kokopu was considered an analogy for lack of significant downstream migration barriers between the site and the ocean. If banded kokopu were absent and the site had seemingly optimum habitat, a downstream migration barrier could have been present.

**Banded kokopu presence:** The final eight sites were selected following confirmation of banded kokopu presence. Kokopu presence was confirmed by night identification under spotlight. Sites were spotlighted using a 10 watt halogen bulb connected to a 6 volt battery, between 2100 and 0400 hours.

**Site Selection Checklist**

Essential attributes:

- Within the vicinity of the Waitakere Ranges
- Banded kokopu present (no downstream migration barriers)
- Within a 15 minute walk from the roadside
- Lack of riparian vegetation draped in the stream
  - Less than 10% developed land in the catchment
- Stony bottomed stream
- Fast enough flow rate to collect drift
- Outside Auckland City water catchment area
- Native riparian vegetation cover over the stream

Preferable attributes:

- Presence of debris dams
- Presence of undercut banks and tree roots
- Pools with shallow, slow moving water
- Between 1-20km from the ocean.
- Roadside site
2.2 SITE DESCRIPTIONS

The vegetation of the Waitakere ranges was milled by loggers from about 1840 to 1940 for timber as a commodity and to clear land for agricultural use. Only small pockets of the ranges were left unmilled, with the rest being subjected to varying intensities of logging over those years (Harvey and Harvey, 2006). Fire events have also historically shaped the nature of the vegetation of the Ranges, as Maori and Europeans cleared land for cultivation. As a result of these historic disturbances to the land, much of the vegetation is recovering and is relatively young (Harvey and Harvey, 2006). The majority of the Waitakere Ranges is now under the protection and management of Auckland Regional Council.

The eight study sites were selected from throughout the Waitakere Ranges, west of Auckland City, and their locations are illustrated in Figure 2-1. The presence of banded kokopu is strongly associated with the presence of riparian vegetation (McDowall, 1990; Rowe et al., 1999) consequently, for sites to have banded kokopu present it was assumed that 50-100% stream cover was required. This study aimed to assess the variation in stream cover, with this 50-100% range being the primary dependant variable.

All study reaches comprised of a 150m length of stream and site photos were taken at a position considered to be representative of each study reach. Photographs were captured using a digital camera with a 167° fish-eye lens, to visually summarise both the in-stream and ex-stream environments at each site as best as possible. Site descriptions describe the general conditions observed at each study reach, but as is inevitable with dynamic environments such as stream systems, there was local and micro-scale habitat variation in and outside the stream throughout each sample reach. Measurements taken along each study reach were averaged to provide a representative data parameter of the average conditions for that 150m of stream.
Figure 2-1  Site location diagram. Site numbers in red are the four sites sampled again during winter. Sites 4, 6, 7 and 8 drain out to the west coast, sites 1, 2, 3 and 5 drain out into the Waitemata Harbour to the north-east.

Figure 1 shows all sites selected for this study. Whakatai, Opal Pool, Ocean View, Karekare and Waitoru Streams flow out across the west coast beaches and into the Tasman Ocean. Swanson, Opanuku and Opanuku Tributary Streams flow in a north-easterly direction to the Hauraki Gulf, then ultimately to the Pacific Ocean. The lower reaches of the latter streams flow through urban and residential zones before reaching the ocean.
2.2.1 OPANUKU STREAM

Figure 2-2  Opanuku Stream Site with visible bank ferns, bank grasses, wheki ponga (*Dicksonia squarrosa*), mahoe (*Melicytus ramiflorus*), rimu (*Dacrydium cupressinum*). Patches of kauri ricker (*Agathis australis*) and kawakawa stands were also evident along the study reach.

The Opanuku Stream Site is located upstream of a small campsite along the Opanuku Pipeline Track, off Mountain Road. 75m along the study reach there is a 1.5m deep pool used as a public swimming hole. There is a large vegetation gap in the canopy above this pool likely caused by human disturbance of the stream bank. At the end of summer, there was a noticeable amount of filamentous algae present in this pool. This pool was considered unrepresentative of the stream reach and was excluded from all analysis; study measurements were taken either side. Benthos composition was consistent along the reach as was primarily composed of small embedded boulders and cobbles.
2.2.2 SWANSON STREAM

Figure 2-3 Swanson Stream Site with visible wheki ponga (*Dicksonia squarrosa*), nikau palm (*Rhopalostylis sapida*), bank grasses and kauri ricker stands (*Agathis australis*).

The Swanson Stream site begins above its intersection with Cassel Stream and continues 20m upstream of a vehicle bridge at the end of Tram Valley Road. Occasionally, small amounts of litter were tossed off this bridge into the river. During drift summer drift sampling three snapper carcasses were found decomposing below the bridge. Although not visible from the stream, there are some low-density residential properties within 75m of the stream. The reach is close to the beginning of the Swanson Pipeline Track and on one occasion, a dog being walked by its owner was observed wading up and down the stream while drift sampling was taking place. The benthos consisted primarily of embedded cobbles and small boulders and was consistent along the reach.
2.2.3 WAITORU FALLS STREAM

Figure 2-4 Waitoru Falls Stream Site showing a small, slow flowing pool section. Visible vegetation is nikau palm (*Rhopalostylis sapida*) and wheki ponga (*Dicksonia squarrosa*) amongst a canopy of primarily kanuka (*Kunzea ericoides*).

Waitoru Falls Stream is named after the toru tree, which is still found in the area (Harvey and Harvey, 2006). Waitoru Falls stream was the study site with the lowest flow, although it contains a number of small localized pools of up to 70cm depth. The benthos is comprised primarily of large sized boulders, with some bedrock. These boulders could act as significant cover, and siltation was evident on boulders in the bottom of pools. The upper end of the reach disappears under large boulders where the flow no longer becomes visible. The upper limit of the reach is delineated by the Waitoru Falls itself, were the water trickles down a benthic rock-face of approximately 10m in height. The lower end of the reach intersects with Stony Creek, a stream which runs alongside Stoneycreek Drive and drains into Waitakere River. There is evidence of some old car bodies next to Waitoru Falls, the cars having been dumped off Bethells Road above. The occasional car component is seen in the stream (one being an old car battery).
2.2.4 KAREKARE STREAM

Figure 2-5 Karekare Stream site. Vegetation visible includes nikau palm trees (*Rhopalostylis sapida*), kiekie (*Freycinetia banksii*) and bank ferns.

Karekare stream is a coastal stream draining a catchment of fully native forest, with the exception of Karekare road that passes through the catchment. The downstream end of the study reach begins above private property at 83 Karekare Road. The benthic substrate consists primarily of boulders, with some cobbles. Approximately 150m upstream of the upper end of the study reach, there was a 10m waterfall. The water became airborne and the benthic rock behind the waterfall was wet, but overhanging. There is a possibility that banded kokopu are not able to scale this waterfall to reside in waters further upstream.
2.2.5 OPANUKU TRIBUTARY STREAM

Figure 2-6  Opanuku Tributary Stream showing a riffle habitat. Visible vegetation is wheki ponga (*Dicksonia squarrosa*), bank grasses, nikau palm (*Rhopalostylis sapida*) and mosses that line the banks.

The confluence of the Opanuku Tributary is 30m upstream of the Opanuku Stream study reach, entering in from the true left. The substrate of this stream consists primarily of cobbles, with some small boulders and an occasional patch of basement rock.
2.2.6 OCEAN VIEW STREAM

Figure 2-7 Ocean View Stream Site. Vegetation visible is nikau palm (*Rhopalostylis sapida*), wheki ponga (*Dicksonia squarrosa*) and Mahoe (*Melicytus ramiflorus*).

The Ocean View Stream site begins at its confluence with Karekare Stream behind McCreadies Paddock Campsite operated by Auckland Regional Council. The benthos consisted primarily of cobbles and small boulders along the reach. The catchment is entirely native with the exception of a minor amount of low density housing along the ridgeline of the upper catchment.
2.2.7 OPAL POOL STREAM

The lower end of the Opal Stream Site is bordered by Lone Kauri Road for the first 50m, causing vegetation to be thin on the true left bank with the road less than 5m from the stream (to the right of Figure 2-8). The reach benthos is mostly sandy gravel, with three aggregations of boulders along the reach. Below the study reach is a series of cascades over boulders and into small pools. There is a minor amount of low density housing and pasture land use in the upper catchment. At 120m along the reach, the stream flows through two large culverts under a driveway, the study reach carries on 30m upstream of these culverts.
2.2.8 WHAKATAI STREAM

Figure 2-9 Whakatai Stream Site, surrounded predominantly by nikau palm (*Rhopalostylis sapida*), with the occasional mahoe (*Melicytus ramiflorus*), kiekie (*Freycinetia banksii*), puriri (*Vitex lucens*), taraire (*Beilschmiedia tarairi*) and some bank grasses.

Whakatai Stream is located at along Whites Track, accessible from North Piha Road. The stream reach begins where Whites Track bisects Whakatai Stream, above its confluence with Marawhara Stream. The stream benthos consists primarily of small boulder and cobbles. The upper end of the study reach terminates below a small (2m) trickling cascade over benthic rock.
2.3 METHODS

The methods below detail five main types of data obtained; water quality, fish habitat, insect abundance, vegetation surveys and fish information.

All parameters were measured once during summer and once during winter, with the exception of vegetation survey methods. Vegetation survey data, stream width and distance to the ocean was only recorded once for each site. This was done because the native vegetation in each of the study reaches is predominantly evergreen and stream cover characteristics are unlikely to change significantly during the year, and are assumed to remain constant throughout this study.

All data was taken at a time when water levels were considered to be at normal ambient flow and not affected by the last rainfall event. Occasionally there were periods of intermittent rainfall that occurred during drift sampling, none of these periods of rainfall caused significant water fluctuations to occur for a prolonged period of time.

2.3.1 WATER QUALITY

All water quality measurements were taken on the same day for each sampling season and within the hour for each sampling site. All measurements were taken at the downstream end of each study reach. Where required, multiple readings were taken at each site to assess equipment and sample preparation variance. These readings were then averaged to give the final water quality results. All readings were taken from areas of flat water in pool situations.

2.3.1.1 pH

The concentration of hydrogen ions is expressed as pH. The pH scale is logarithmic and is measured from 1-14, the smaller value representing a larger hydrogen ion activity and increasing acidity. Acidic waters can increase the solubility of toxic to toxic metals (Dodds, 2002). The range normally found in aquatic systems is around neutrality at pH 7.

Measurements were taken in the field with a handheld Mettler Toledo, MP120 pH meter. This meter had been calibrated with pH 4 and pH 7 laboratory standards to ensure they were reading within an acceptable margin of error, before being taken into the field. The sensor was left in the water for two minutes to settle before readings were recorded. Two pH readings were taken at the same position, to assess reading variance of the meter. The final pH was recorded as an average of those two readings.
2.3.1.2 DISSOLVED OXYGEN

The amount of oxygen dissolved in water is primarily a function of water temperature, but also atmospheric pressure, diffusion and metabolic activity rates (Dodds, 2002). It can be measured in percentage saturation, from 0-100%. Dissolved oxygen is important for respiration of in-stream fauna (Brown, 1987).

Measurements were taken in the field with a handheld Mettler Toledo, M0128 Dissolved Oxygen meter. This meter also measures water temperature, as the capability of water to retain dissolved oxygen is reliant on water temperature. The sensor was left in the water for two minutes to settle before readings were taken. Two dissolved oxygen readings were recorded at the same position, to assess reading variance of the meter. The final dissolved oxygen result is taken as an average of those two readings.

2.3.1.3 CONDUCTIVITY

Conductivity simply measures the relative amount of electricity that can be conducted by water, showing concentration of dissolved ions present (Dodds, 2002). Conductivity can also be used as a surrogate for water salinity content (Jeffries and Mills, 1990).

Measurements were taken in the field with an EDT Instruments, GP383 conductivity meter. A manual calibration was performed at each site before readings were taken. The calibration consisted of:

- Temperature calibration measurements taken from the temperature reading on the dissolved oxygen meter. The water temperature adjustment dial on the conductivity meter was then set to the stream temperature.
- The default meter conductivity temperature compensation rate of 2%/°C was set and not altered between sites.
- A laboratory conductivity standard of 1413 µs/cm was used and the calibration dial was adjusted until the equipment was reading this measurement.

Two readings were then recorded at each site to assess reading variance of the meter. After calibration, it was found that there was no difference between replicate readings from the meter. As there was no error between readings, only one measurement was required from each site.
2.3.1.4 TURBIDITY

Turbidity is a measure of solids that are suspended in water and their transport is dependent on water velocity and turbulence. Streams become turbid after rain events and silt can cause some difficulty for some species of fish and invertebrates (Dodds, 2002).

A potte of water was taken from each stream reach. Care was taken to ensure this sample was collected from an undisturbed area of the stream. This potte was identifiable by placing a waterproof paper label inside with the date, site name and “Turbidity Sample” written on with pencil. All samples were kept cool in an insulated bin during transport and then stored in the laboratory refrigerator at 6.5°C. Turbidity samples were processed within the next 20 hours in the laboratory. This was done using a HACH model 46500, 2100p Portable Turbidimeter. This meter calculates the ratio of scattered light received at 90°, to the proportion of light transmitted through the sample and gives a final reading in Nephelometric Turbidity Units (NTU).

Each water sample potte was agitated prior to sub-sampling to re-suspend any sediment particles that had settled. The approved turbidimeter glass vials were rinsed with water from each site to ensure that no residual sample from a previous site remained, before being filled with sample water. Care was taken to ensure the vial was polished clean, leaving no water or fingerprints on the glass surface. Three readings were taken of each sample to assess reading variation. A further three readings were taken from each sample using a second vial, to assess reading variation between vials. All readings were then averaged to give a NTU turbidity reading.

2.3.1.5 TEMPERATURE

Temperature readings were initially taken with a mercury thermometer; however readings were too coarse, only being visually defined to the nearest 0.5 °C. The final temperature readings were taken at the same time as the other water quality readings and were recorded from the dissolved oxygen meter. The two temperature readings were then averaged to give the site water temperature reading.

2.3.1.6 STREAM DISCHARGE

Stream discharge measurements were calculated utilising wetted width, depth and water column velocity measurements. Water depths from the water surface to the substrate were taken at 10cm intervals across the stream. At the mid-point between each depth measurement, a water velocity measurement was taken with a Marshall-McBirney water current velocity meter (Figure 2-10). Each measurement was taken at 0.4x depth from the benthos (Allan, 1995). A cross-sectional area was then calculated for each 10 cm segment across the stream and multiplied by each corresponding water velocity measurement to give a discharge measurement for each stream segment. All segment
discharges were added to give a total discharge for the stream. A minimum of three discharge measurements were taken at each stream to approximate the error involved in discharge estimation. Calculated discharge measurements appeared to agree with relative discharges observed visually.

Figure 2-10 Marshall-McBirney water current velocity meter, showing a measurement of 0.24 m/s\(^{-1}\).

2.3.2 INSECT ABUNDANCE

Insect abundance and activity in the stream was measured by two methods. One sampled terrestrial and macroinvertebrate drift activity, the other sampled macroinvertebrate abundance. All insects were identified with the help of Winterbourn, Gregson et al. (2000), Crowe (2002), Gray and Zollhoeffer (2006) and macroinvertebrate identification quick-guides provided by NIWA (2007).

2.3.2.1 SURFACE AND SUBSURFACE DRIFT ACTIVITY

The drift sampling method utilised in this study sampled stream drift from the whole water column, from surface to benthos. This was intended to collect both allochthonous and autochthonous insect material being transported in the water column.

The samplers were constructed similar to the description detailed by Field-Dodgson (1985) and are shown in Figure 2-12. The samplers utilise a Marley PVC downpipe adaptor (MC-141), traditionally utilised for rainwater guttering systems. This adaptor has an aperture of 105x35 mm that followed through to a circular 80 mm diameter opening. Attached to the circular end of the piping adaptor is a 1m long ‘sock’ constructed of 500 µm mesh, fastened to the adaptor with a hose clamp. At the end of this sock is glued a screw type plastic attachment, to which a sampling pottle can be connected. At the bottom of this pottle is a 20mm hole, sealed with 500µm mesh to allow water to flow through the pottle, gathering sample contents at the bottom.
The sampler was placed either lengthwise (long edge parallel with the substrate) or vertically (short edge parallel with the substrate) in the water column, dependant on the flow velocity being sampled. If the flow was high velocity, the sampler was positioned vertically to ensure samples were not too large as to be difficult to process in the laboratory. The sampler could also be placed vertically in the water column if it was difficult to find a shallow depth of water in the appropriate position along the stream reach. Conversely, if flow velocity was slow and the stream was shallow, the sampler could be placed horizontally to sample more of the streams surface, as shown in Figure 2-11 and Figure 2-12.

Figure 2-11  Drift sampler viewed looking downstream. Note: opening adaptor is orientated with a horizontal aperture.
Initially, samplers were bolted to two warratah stakes that were driven into the substrate for stabilisation. This option was abandoned, primarily because the damage caused to the stream benthos when removing stakes was not necessary considering the short amount of time they were required in the stream. Additionally, carrying the stakes and sledgehammer into sites required some difficulty. Alternatively, each sampling unit was bolted to a length of steel and strategically placed across the stream. This technique proved to be easier, as the steel was simply supported and weighted down with rocks at either end.

Replicate sampler at each site reduced the error presented by lines of preferential drift flow and micro-scale differences in stream and terrestrial invertebrate activity along the reach. The samplers were placed in locations along the stream reach where there was no filter type constriction of the water surface immediately upstream, such as a debris dam. They were also placed where water was between 20 and 100 mm deep and the flow had enough downstream motion for its surface not to be affected by wind flowing back upstream. Preferably this occurred in a ‘run’ type situation.

Samplers were placed immediately downstream of an area of riffles. This strategic placement was elected for to reduce preferential feeding effects by banded kokopu while drift sampling was taking place. This is based on the assumption that banded kokopu would not be actively feeding in riffle...
areas. Selective consumption of insects that had become entrained in drift before they reached samplers could cause an under-representation of preferred drift species.

Care was taken to avoid leaving samplers in streams during predicted high rainfall periods. This was done to avoid flood effects on drift composition, ensure they were sampling a consistent amount of flow and to prevent samplers getting washed away. Depth measurements were written on the side of each sampler and the submerged aperture was recorded for each individual sampler at each different location. A water velocity reading was also taken at the sampler entrance. Sampler aperture and water velocity measurements can be multiplied to calculate a water discharge volume measurement for each sampler, for each day or night drift period.

Sample collection involved lifting each sampler and flushing down the mesh with stream water, ensuring all trapped contents were flushed down into the terminal pottle. The pottle was then unscrewed, removed and its contents flushed out and preserved with 75% ethanol (Stark et al., 2001). The ethanol was ejected from a 60ml syringe and a 7cm hypodermic needle, and drift contents flushed into a plastic sealed storage pottle. A waterproof label was placed inside each pottle with the sample, detailing: site name, date, “Day” or “Night” sample, “Drift”, and replicate sampler number (1, 2 or 3).

Each drift sample was stored in the laboratory until it could be processed. Each sample was flushed out of the pottle with water onto a 500 µm mesh, taking care to flush all contents off the inside of the lid, the label and the inside of each pottle. The contents were sorted on a glass tray under a 10x binocular dissection microscope. The waterproof label was transferred into a 25ml plastic pottle half filled with 75% ethanol and each insect taken from the sample was then placed in this pottle for later identification.

Leaf matter from each sample was placed onto a small 1g aluminium tray and air dried for a period of approximately 48hrs in the laboratory fume hood. The leaf matter was then weighed to the nearest 0.01 g, with a subjective note as to the general leaf matter composition. Twenty aluminium trays were weighed and found to be 1 g ±0.01 g; therefore a 1 g standard measurement was subtracted from each leaf matter weight to account for the weight of the aluminium tray. Large leaves over 5 cm in length were rinsed down and excluded from analysis, as these made samples too large and leaves of this size were often discarded in the field.

Some species of insect have both aquatic and terrestrial phases of their cycle. For the purposes of this study, aquatic insects that have a terrestrial life cycle phase were recorded as being terrestrial if they were captured in that phase. Also, some species are able to walk on the water’s surface and although they are not technically trapped in water tension, they can contribute to fish diet by the
same mode as terrestrial insects do, from above water's surface. For this reason, insects such as the Hemiptera order of water bugs have been classified as being terrestrial.

Raw counts of drift activity for each sampler were normalised by dividing count data by the amount of water that flowed through each sampler for each day or night period. Although the exact time for collection of drift samples varied due to distance between samplers and between sites, they were generally left out for 14 hours during the day time and 10 hours during the night time. The discharge for each sampler was calculated for each 14 or 10 hour period and captured drift data was divided by this discharge. This gave a drift rate per m³ water for each sampler during each day or night period, normalising for varied submersion depths of sampler apertures and different flow rates for each sampler. The data was then pooled over three days and three nights for each site where drift activity for day or night periods are presented.

2.3.2.2 MACROINVERTEBRATE ACTIVITY

Kick sampling estimated stream benthic macroinvertebrate abundance as a diet opportunity for Galaxias fasciatus at each site. Protocols laid out in Stark, Boothroyd et al. (2001) were followed. Kick sampling utilised a 29x29x29 cm triangular shaped kick net with 250 μm mesh. Each kick sample was undertaken in cobble substrate, in a riffle location as the most productive microhabitat. The flat side of the triangle net was placed on the substrate and a vigorous kicking motion undertaken on the substrate within a 40 cm radius upstream of the kick net. The disturbed detritus and macroinvertebrate components are then washed into the kick net with stream flow.

The procedure was repeated three times, each at a different location along the study reach, for 20 seconds each. The three sub-samples were combined and rinsed into the bottom corner of the net. The net was then inverted to empty its contents into a sample pot. The pot was then preserved with 75% concentration ethanol. A label was placed in each pot detailing: site name, date, “Day” or “Night” sample, and “Kick”.

These samples were stored in the laboratory and macroinvertebrates sorted out from detritus for later identification. The macroinvertebrates were placed in a 25 ml pot half filled with 75% ethanol and the same waterproof kick sample label placed inside. The leftover detritus material was discarded.
2.3.3 IN-STREAM HABITAT ASSESSMENT

2.3.3.1 DEBRIS DAMS, BANK COVER AND BANK SCARRING

The extent of potential in-stream habitat for banded kokopu was measured by extending a tape measure along subjectively delineated areas of debris dams, bank cover and along bank scars of true left and true right banks. Measurements were recorded in meters.

Debris dams were chosen combinations of non living vegetation that had become snagged (Figure 2-13). These commonly included twigs, leaves and occasionally fallen branches. It did not include debris dams that were left outside the stream wetted area by flooding and were not submerged in water. Debris dams were calculated as a proportion of the reach length covered by debris dams.

![An example of a structurally complex debris dam and a bank scar. Photo taken looking down Whakatai Stream. Four banded kokopu were captured from this debris dam during winter sampling.](image)

Undercut banks were areas along the stream margins where water extended underneath the bank and the underside could not be seen with the naked eye (Figure 2-14). These areas included overhanging banks and areas of tree roots and root mats extending into the stream from the surrounding riparian vegetation. Bank cover was calculated as a proportion of both true left and true right bank lengths combined (i.e., stream reach length x2).
Figure 2-14 An example of undercut bank habitat in Opal Pool Stream, showing the darker light environment and some minor root cover.

Bank scarring measurements were taken along each bank as a measure of disturbance. Scarring was defined as areas of bank at stream level that showed visible signs of recent damage, often showing the underlying clay soil (Figure 2-13). Bank scarring was recorded in meters and as a proportion of both true left and true right banks lengths (i.e., stream reach length x2).

2.3.4 ELECTRIC FISHING

A portable battery powered NIWA Instrument Systems Electric Fishing Machine (EFM300) backpack was utilised for electric fishing. This equipment supplied a DC voltage electric current to the water and through a wand (anode). This stunned freshwater fish in the near vicinity of the wand for capture in a small hand net and also allowed their subsequent return to the stream in an unharmed state. If the water was conductive due to large amounts of salts and minerals, then backpack voltage was increased to induce a larger field of inference around the electric probe. No current over 200 volts was required as water quality at all sites was particularly good and there were no occasions where fishing occurred after significant rain events (which would increase water conductivity).

The same user handled the electric fishing machine for each site and sampling effort was consistent on the first pass. Electric fishing began at the bottom of each stream reach and involved slowly
walking up the stream, sequentially probing through the water column, watching for any sign of banded kokopu. Extra time and effort was taken to probe in and around likely fish habitat such as debris dams, undercut banks and root cover. Areas of fast flowing water such as ripples and large deep pools were not given so much attention as it is known that banded kokopu do not often reside in these habitats. Once stunned, fish were captured with a small hand net and placed in a container of fresh stream water with a lid on to prevent them jumping or climbing out.

Any other stunned fish species were noted as species present, but not captured or counted. Only unusual fish species such as koaro (*Galaxias brevipinnis*) or shortjaw kokopu (*Galaxias postvectis*) were captured, measured for size and released. Species present information will be useful for contribution to the New Zealand Freshwater Fish Database (NIWA, 2008), along with more specific length, weight and condition information for banded kokopu from each stream reach.

A stream with an abundance of undercut banks, debris dams and root cover required a long amount of time to effectively fish. Electric fishing 150m of stream required 3-7 hours to fish effectively, including fish treatment time. Due to the large amount of time it took to fish out stream reaches with relatively high densities of fish, it was not always possible to complete a second electric fishing pass of the reach. If a stream did not contain high densities of fish, then extra time could be taken to subjectively fish out areas of habitat a second time if they were thought to contain more fish than was initially captured. As this study is not primarily concerned with fish density, this subjective second pass was often utilised to provide as many fish results as practically possible for each site.

Fish were individually taken out of the containing bucket and placed in a bucket of stream water and fish anaesthetic, at a prescribed dosage of 0.05ml Aqui-s™/5L water. Placing fish individually into the bucket containing anaesthetic allowed their level of sedation to be individually monitored. Each fish was removed when it reached a satisfactory state of sedation, described as being when the fish looses equilibrium and does not react to being handled (AQUI-S New Zealand Ltd). It was important to ensure sufficient anaesthesia because if a fish reacted by way of muscular movement when the gut flushing apparatus was inserted into its mouth and down its gullet, it could damage internal organs.

Each fish was weighed with a Denver balance to the nearest 0.01 g and fish length measured to the nearest millimetre using a ruler. This information was recorded on waterproof paper with site name, fish identifier, date, weather, site notes, other fish species observed and any notes about each individual fish captured. This fish survey data will also contribute to the New Zealand Freshwater Fish database (NIWA, 2008).

A 60ml syringe attached to a 14G*3 1/4” (2*80mm) blunt tip hypodermic needle is filled with stream water. This needle was gently inserted into the fish mouth and into its gullet. The fish was carefully
inverted and water from the syringe was flushed into its gullet, causing the gut contents to then be
flushed out of its mouth and into a small plastic tray. Fish under 150mm in length were flushed with
two syringes full of water; fish under 150mm length were flushed only once.

Those gut contents were then poured into a pottle, the bottom of which had a 500 μm mesh to sieve
the gut contents out from water (sieve pottles are the same pottles used for drift samplers). A
separate syringe filled with ethanol was used to then flush those contents out of the sieve pottle and
into a 25 ml storage pottle. Each gut content sample was accompanied with a waterproof label
stating; date, site name and fish identifier (to relate to weight and length results) written on each
label in pencil.

2.3.5 VEGETATION COVER SURVEY

Riparian vegetation characteristics are measured using cover estimates. Because of the fine detail
required to estimate cover for the purposes of this study, three different methods of cover
estimation were utilised and are presented below.

2.3.5.1 Densiometer

A subjective method of vegetation cover was estimated utilising a Model A Spherical Convex
Densiometer (Lemmon, 1957) as seen in Figure 2-15. This equipment has convex, spherical mirror
that reflects the above canopy when held horizontal, according to a small mounted spirit level.
There is a grid of squares on the mirror face and measurements are taken by counting the grid
intersections not covered by vegetation.

The standard procedure has been modified according to Strikler (1959) by placing masking tape
over the mirrored surface of the densiometer in the orientation shown in Figure 2-16. Utilising this
modification visually excludes the user from cover readings. A standard height of 30cm off the stream surface reduces recording variation between people of different heights and includes the contribution of low hanging vegetation to stream cover. Because of this, the Strikler method is ideal for gaining estimates of cover over stream systems. Bank measurements can be analysed separately to stream middle measurements, to assess how riparian cover varies across the transect.

Figure 2-16 provides an example of cover estimation from a projected hemispherical image of the above vegetation canopy onto the mirrored surface. Canopy cover was projected on 9 out of the 17 crosses in the image; therefore this image represented 52.9% cover.

The densiometer orientation and transect method is illustrated in Figure 2-17. A transect across the stream was recorded every 6.6 m along the stream reach. This distance was chosen to provide adequate quantities of data to estimate cover over each stream reach to a fine enough resolution required for this study. At each transect along the stream reach, six measurements were taken, each at 30 cm above the stream surface. They were taken facing: true left bank, upstream middle, true right middle, downstream middle, true left middle, and true right bank. This is illustrated in Figure 2-17. Each bank reading was taken at 30 cm distance from the bank.
DIFN is the measure of light received on a horizontal plane as a proportion of that from an unobstructed, uniform sky (Davies-Colley and Payne, 1998; Boothroyd et al., 2004). This stream cover parameter was recorded using the Li-cor™ LAI-2000 Plant Canopy Analyser.

This equipment utilises a fisheye lens with a 74° field of view from the zenith (Figure 2-18). The light is transposed onto five annular light detectors as shown in the Figure 2-18 detector and Figure 2-19b. This transposes an image of the upper hemisphere (canopy) on the circular sensors and represents a range of angles from the zenith, recording a DIFN reading from each of the five sensors. These five sensors are then averaged to give a DIFN cover reading for that particular moment in time.
Light readings were recorded under preferably isotropic (uniform) light conditions, such as those present at early morning (before sunrise) or late evening (after sunset). Any direct open sky light conditions that reflect off vegetation can bias cover estimates with the analyser, by viewing those reflections as open sky. A blue filter is present in the analyser that reduces the chances of this cover miscalculation. This is done by rejecting light above 490 nm, typical of that spectrum reflected off foliage. Lens viewing caps are also fitted to the fisheye lens to eliminate bias caused by the user blocking light from reaching the sensor and to reduce the chance that sunlight will shine directly into the lens. By fitting a 180° view cap to the end of the sensor (Figure 2-19a) and orientating it in a Southerly direction, essentially the user and any direct sunlight are blocked from the lens’s view and only canopy gaps viewed to the southerly orientation are recorded.

Figure 2-18 The light canopy analyser lens, in diagrammatic profile (LI-COR Industries, 1992).

Figure 2-19 Figure a): fish eye lens of the plant canopy analyser, fitted with a 180° viewing cap and a plastic bottle underneath attached with plastic zip-ties to prevent immersion. Figure b): canopy area viewed through the fish-eye lens that each of the five annular rings will record data from (NB: figure b) is without a 180 degree view cap fitted; LI-COR Industries, 1992).
Paired sensors were used for this method. One was a reference sensor (A logger) placed in open space with no cover obstructions such as an open field, to measure ambient light conditions. This sensor was located as close as practically possible to the stream location and set to log light conditions every 30 seconds. The other light sensor (B logger) was taken under the canopy and readings were manually logged every time a trigger button was pressed. Both A and B sensors were orientated to a compass direction and adjusted to the horizontal by a mounted spirit level to ensure they were both measuring the same portion of upper hemisphere sky.

Ten readings were taken along the study reach. This occurred twice, once at stream level and once at bankfull height, giving a total of 20 readings for each stream reach. For a 150m long reach, readings were taken at 7.5m spacing, the first reading being taken at the zero metre baseline. Readings were taken at 10, 30, 50, 70 and 90% of the stream width and were always orientated in a southerly direction. This was to ensure the 180° viewing cap was always blocking direct light from the sun (opposite for measurement in the northern hemisphere).

Later, the two measurements were aligned using the Li-cor™ Data File Viewer software (LI-COR Industries, 2008). Every time a B log reading was taken, the closest A log reading was associated with it as a representation of open sky light at that time. This pairing was done to normalise readings to account for variation in sky lighting conditions that occur over the course of the day. The Li-cor™ software can then delineate B log readings from A log readings and calculate an actual DIFN cover measurement for each study reach.

![Recording method of Light Canopy Analyser, taking readings every 7.5 m up a 150 m reach. Readings taken at 10, 30, 50, 70 and 90% of the stream width, orientated in a southerly direction.](image)
2.3.5.3 HEMISPHERICAL PHOTOGRAPHY

A description of stream cover obtained by the light canopy analyser gave overall cover measurements for each site, but where this cover came from can be further defined. When considering riparian vegetation insect contribution to a stream, ‘all cover is not created equal’ (PersComm Davies-Colley, 2008). Vegetation cover over streams will be differentiated into ‘near’ and ‘far’ proximity cover categories, attempting to measure their differential contributions of insects to the stream.

Near cover, by definition, will aim to quantify how much vegetation is close to or overhangs each stream. Its adjacent proximity to the stream may enhance its ability to contribute relatively large numbers of insects to the stream per total leaf area of the plant. ‘Far’ cover will primarily measure tree canopy that is viewable from the stream, but not immediate to the stream, such as the crowns of large canopy trees. Far cover may not contribute a large proportion of insects per leaf area of each plant, but may contribute insects to the stream from a wider collection area.

This method involved the use of a Canon 450D Digital Single Lens Reflex (DSLR) camera. The camera had a Sigma 10 mm 1:2.8 EX DC Fisheye HSM lens attached, which gave dedicated fisheye 167° angle of view. A dedicated fisheye lens reduces the amount of barrel distortion that may occur when compared to a fish eye lens that is attached to a normal lens. An ideal lens for this type of canopy analysis will have 180° field of view and be able to take photos of the whole upper hemisphere and vegetation in close proximity to the stream, however a lens giving a 167° viewing angle was the only one available for this study.

Table 2-1  Showing the distances separating each photo taken along the study reach.

<table>
<thead>
<tr>
<th>Photo Number</th>
<th>1</th>
<th>2</th>
<th>3</th>
<th>4</th>
<th>5</th>
<th>6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Distance along Reach</td>
<td>12.5</td>
<td>37.5</td>
<td>62.5</td>
<td>87.5</td>
<td>112.5</td>
<td>137.5</td>
</tr>
</tbody>
</table>

The camera was mounted on a tripod in the middle of the stream, as close to the water level as possible, with the lens facing vertically to capture the canopy. This is often not possible given the height of the tripod and variable stream depth; consequently most photographs were taken from approximately 0.5 m off the waters’ surface. Given that the camera lens did not give a true 180° hemispherical image and it was important to capture as much close proximity vegetation as practically possible, the camera was orientated so the top of the image (and the top of the camera) was facing downstream for each photo. This enables the long axis of each rectangular photograph to take in as much close proximity vegetation along the stream banks as possible. Photos were taken
with a set manual focus close to infinity, using the aperture priority function set on an F-stop of 5.6, with camera auto-adjustment of shutter speed.

Photos were taken at six points along the study reach, the first point being 12.5 m from the beginning of the reach and each point thereafter being taken at 25 m intervals. This distance between photos reduces the chance of photo overlap and pseudo-replication of results with six photos being taken for stream reach. At each point, three photos were taken with auto-bracketing to ± 1 exposure stop, in case further computer analysis was required at a later stage to delineate vegetation boundaries.

All photos were uploaded to computer and the medium exposure shots were processed by Adobe Photoshop® to flatten each image to greyscale. Image histogram data was then exported to excel utilising the Reindeer Graphics Fovea Pro® plug-in for Adobe Photoshop®.

![Example of a hemispherical photograph of the vegetation canopy above a stream, with colour information discarded and flattened to grey-scale to represent 255 unique shades of grey.](image)

Figure 2-21  Example of a hemispherical photograph of the vegetation canopy above a stream, with colour information discarded and flattened to grey-scale to represent 255 unique shades of grey.

Once cover has been isolated into categories of ‘near’, ‘far’ and ‘open’, Photoshop® can calculate the fractions of these cover categories from the whole, as illustrated in Figure 2-23. Although reach vegetation cover was determined by the light canopy analyser, this method was intended to provide information on the characteristics of that cover.
Figure 2.22 A light distribution diagram for the grey-scale hemispherical photograph in Figure 2.21. Image is of vegetation canopy over a stream separated into: Near cover 1-15, Far cover 16-215, and Open Sky 216-255 luminescence values.

Figure 2.23 Processed in Adobe Photoshop® to separate out three pixel cover fractions from the original greyscale photo in Figure 2.21. Grey denotes the close cover fraction (threshold 1-15), white denotes the far cover fraction (threshold 16-215) and blue denotes the sky fraction (threshold 216-255). Vegetation fractions are then calculated as a percentage from the whole.
2.3.6 STATISTICAL ANALYSIS

Microsoft® Office Excel® was used to store, investigate and create graphs from all quantitative numerical data collected. The primary statistical techniques used in this study are presented below. Linear regression graphs are presented with correlation data to graphically represent relationships between dependant and independent variables.

2.3.6.1 SIGNIFICANCE TESTING

Mann-Whitney U tests were utilised as a form of non-parametric testing of statistical significance between resultant groups of data, to determine if two sets of data come from the same or different distributions. The advantage of utilizing this test over others is that it does not require the data to be normally distributed. To qualify for this analysis, the data must come from two independent samples and must be ordinal. It is the non-parametric equivalent of the parametric Student’s two sample t-test.

2.3.6.2 CORRELATION ANALYSIS

Pearson correlation data presented in tables throughout this section are derived from multiple simple correlations. This statistical test is designed to show the degree to which two variables ‘move together’. This generated a correlation matrix of r values showing the strength of the each correlation: +1= a perfect positive correlation, 0= no correlation and -1= a perfect negative correlation. Accompanying the correlation value is a correlation significance p value and a t-statistic. Tests were significant if they returned a p-value smaller than 0.05, giving us 95% confidence that the two samples came from different populations. Differences were highly significant if they returned a p-value of 0.01, giving us 99% confidence that the two samples came from different populations.

2.3.6.3 MULTIVARIATE ANALYSIS

Multivariate analysis attempts to reduce a complex set of high-dimensional data into a low-dimensional display to describe community patterns and any relationships that are exhibited (Clarke and Warwick, 2001). This allows data obtained from each sample or site to be assessed for its unique species composition and distribution, based on the extent to which two or more sites share particular attributes. A-priori similarity can be observed by assessing similarities or differences across the samples and comparing these with known or hypothesized inter-relations. All multivariate analysis were undertaken, and illustrations were created utilising PC-ORD software (McCune and Mefford, 1999).

Relationships between samples are shown using a hierarchical dendogram that clusters samples containing similar species. An ordination diagram is presented in unison with each dendogram, to
represent sample or site similarities, presenting similarities in species composition between samples or sites. Cluster groupings were defined at a percentage similarity that best represented relationship groupings in both the dendogram and histogram.

To create each dendogram or ordination diagram, environmental or species data was tabulated into two way data matrices representing the variable for each site. Environmental variable replicates were averaged (to prevent pseudo-replication between replicates) for the eight sites and both seasons, to give 12 site/seasons. Sites within seasons were independent but sites between seasons were dependant. Despite this, both sites and seasons were analysed in the same data matrix to assess both the inter-site and inter-season differences. This indicates the extent to which community niche variation was the result of site (spatial) independence or from seasonal (temporal) independence. Summer and winter seasons are also presented independently for fish gut content analysis in chapter 4.

Cluster analysis created dendograms using the Sorensen (Bray-Curtis) coefficient, with group averaging linkage method. The Non-metric Multidimensional Scaling (NMS) technique was used for ordination, utilizing the Sorenson (Bray-Curtis) coefficient. The major strength of this technique over others is that it:

- Makes no assumptions about the linearity of the dataset (Clarke, 1993; McCune and Grace, 2002)
- Copes with datasets containing a large number of zeros, for example: where species do not appear at a site (McCune and Grace, 2002)
- Is less sensitive to outliers, irregularly spaced observations along gradients or a moderate level of nose in the data (McGarigal et al., 2000).

The NMS is undertaken with 40 re-runs of real data, and 50 re-runs of randomized data, comparing the stress from each to evaluate the strength of the multivariate output, determining whether the observed minimum stress is smaller than that expected by chance.

2.3.6.4 IVELV’S ELECTIVITY INDEX OF PREY SELECTION

The calculation of banded kokopu diet preferences uses Ivlev’s electivity index (Lechowicz, 1982):

\[
Electivity = \frac{(r - p)}{(r + p)}
\]

Where \( r \) = the percent of prey in fish diet and \( p \) = percent of the prey in the environment.
Chapter 3 | The relationship between stream cover and stream insect activity

The relationship between stream cover and stream insect activity

This section outlines and interprets findings from stream cover and terrestrial insect capture methods to answer the chapter objective:

*Determine how the extent of riparian vegetation cover influences the influx of terrestrially derived invertebrate contributions to the stream system.*

This will provide information to test the hypothesis that:

*An increase in vegetation cover over the stream will cause an increase in the amount of terrestrial invertebrates available in stream drift.*

The first sets of results presented are ‘snapshot’ surveys, showing environmental data taken at each of the sites. This snapshot is to provide a picture of environmental conditions between each of the sites and between each of the seasons. Water quality variables, stream habitat and vegetation cover characteristics are then detailed to further characterise sites. Insect collection results are presented to gain an understanding of the relative differences between sites, between day and night periods and finally, between terrestrial and aquatic invertebrate components that comprise insects in stream drift.

Summary data is presented and its structure and significance between sites and seasons is assessed. Significant correlations with other variables are indicated. To provide information of association and relationships between the key dependant (cover) and independent (insects in drift) variables, a number of relationship graphs are presented to graphically assess the strength and degree of interconnectedness between these parameters, with their associated strength and relationship values. These graphs and significance tests provide the analytical foundation to discuss and ultimately answer the objectives of this chapter: does the amount of stream cover affect insect contributions to the stream?
3.1 SITE CHARACTERISATION RESULTS

Site characterization data is presented below to provide comparison between major environmental parameters observed. They provide data to give each stream relative context, aid commentary and provide comparative data for stream cover and insect drift relationship results.

3.1.1 WATER QUALITY

While it is expected that water quality variables continually changed over the course of the study, water quality data was recorded only once for each season, to compare relative differences between sites at one point in time. Summer water quality parameters were taken on 20/4/08 and winter water quality parameters were taken on 26/8/08. This snapshot survey of water quality results will show relative differences in water quality variables between sites (spatial) and between seasons (temporal).

3.1.1.1 pH

pH ranged from 6.95 at Opanuku to a high value of 7.42 at Whakatai Tributary from the summer sampling sites, while pH ranged between 6.46 to a high value of 6.79 at both Whakatai and Opanuku Tributary during winter sampling (Figure 3-1). Mean pH during summer was 7.2, compared to a winter mean of 6.7 at those same sites. Mann-Whitney test showed this to be a significant difference between the summer and winter pH levels (n=4, p=0.03).

An outlier pH reading of 10.6 was recorded at Opanuku Tributary during summer sampling. This was inconsistent with the second measurement taken and with the average readings taken at the other sites, therefore it was removed from analysis and is not represented in Figure 3-1.
Chapter 3 | The relationship between stream cover and stream insect activity

Figure 3-1  Mean stream pH. Error bars = ±1 standard error (n=2). Note: only one reading was taken at Opanuku Trib during summer.

There was no correlation pattern between the four summer and four winter sites (n=4, r=0.166, p=0.884). Correlation analysis results are presented in Table 3-1. Different pH levels tend to be correlated with other types of water chemistry in local geographic regions (Whitton, 1975) and this is possibly evident in relationships with stream discharge, turbidity and conductivity.

Table 3-1  Pearson correlations between pH and other environmental variables. n=12 are summer and winter sites, n=8 are summer sites only.

<table>
<thead>
<tr>
<th>p</th>
<th>Bank Scarring</th>
<th>Turbidity</th>
<th>Discharge</th>
<th>Debris Dams</th>
<th>Conductivity</th>
<th>Stream DIFN</th>
<th>Bank DIFN</th>
<th>Dist to Ocean</th>
</tr>
</thead>
<tbody>
<tr>
<td>r</td>
<td>-0.67</td>
<td>-0.62</td>
<td>-0.62</td>
<td>0.79</td>
<td>0.83</td>
<td>-0.80</td>
<td>-0.78</td>
<td>-0.87</td>
</tr>
<tr>
<td>n</td>
<td>12</td>
<td>12</td>
<td>12</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>

3.1.1.2 DISSOLVED OXYGEN

The lowest dissolved oxygen reading (83.3%) was recorded at Ocean View Stream while the highest was recorded at Opanuku Tributary (106.1%; Figure 3-2). Dissolved oxygen was higher at three of the four winter sites, with only Opanuku Trib recording a higher reading in summer. Mean dissolved oxygen during winter was 90.3% while the mean during summer at those four sites was 91.2%. There was however no significant difference in dissolved oxygen concentrations between summer and winter at the four sites (n=4, p=0.34). Small streams with cool water and riffle habitats, such as
the ones utilised in the present study, can maintain dissolved oxygen concentrations at near saturation (Allan, 1995).

![Figure 3-2: Mean stream Dissolved Oxygen (n=2). Error bars = ±1 standard error.](image)

Correlation analysis showed a relationship between decreasing dissolved oxygen concentrations with increasing temperatures (Table 3-2). This relationship is consistent with the commonly understood principle that these two variables are inversely related (Allan, 1995; Hauer and Lamberti, 1996; Dodds, 2002).

![Table 3-2: Pearson correlation results between dissolved oxygen and other stream variables. n=12 included summer and winter sites, n=8 are summer sites.](image)

<table>
<thead>
<tr>
<th></th>
<th>Conductivity</th>
<th>Temperature</th>
<th>Conductivity</th>
<th>Temperature</th>
<th>pH</th>
<th>Distance to the Ocean</th>
</tr>
</thead>
<tbody>
<tr>
<td>p</td>
<td>0.03</td>
<td>0.04</td>
<td>0.04</td>
<td>0.03</td>
<td>0.05</td>
<td>0.02</td>
</tr>
<tr>
<td>r</td>
<td>-0.63</td>
<td>-0.60</td>
<td>-0.74</td>
<td>-0.78</td>
<td>-0.70</td>
<td>0.79</td>
</tr>
<tr>
<td>n</td>
<td>12</td>
<td>12</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>

3.1.1.3 CONDUCTIVITY

Two conductivity readings were recorded at each location, consistently showing no difference between each reading. The highest conductivity reading was recorded at Ocean View Stream during summer (465 µs/cm), while the lowest was recorded from Opanuku Tributary during winter sampling (116 µs/cm; Figure 3-3). The mean conductivity from the winter was 227.8 µs/cm,
compared to 337.8 µs/cm from those same sites during summer. There was no significant difference between summer and winter mean conductivity levels ($n=4, p=0.49$). Lower conductivity readings at each of the winter sites compared to the summer sites may be the result of increased stream discharges during winter causing a diffusion of airborne ocean salts that are flushed from surfaces and end up in surface or sub-surface drainage flow.

The correlation between increasing conductivity and decreasing distance to the ocean (Table 3-3) may be caused by airborne salts from the ocean, causing higher conductivity readings at sites in close proximity to the ocean.

Table 3-3 Pearson correlations between conductivity and other stream environmental variables. $n=12$ included summer and winter sites, $n=8$ are summer sites, $n=4$ are winter sites.

<table>
<thead>
<tr>
<th></th>
<th>Debris Dams</th>
<th>Dissolved Oxygen</th>
<th>Debris Dams</th>
<th>pH</th>
<th>Dissolved Oxygen</th>
<th>DIFN Stream</th>
<th>DIFN Bank</th>
<th>Dist to Ocean</th>
<th>Temperature</th>
</tr>
</thead>
<tbody>
<tr>
<td>$p$</td>
<td>0.00</td>
<td>0.03</td>
<td>0.01</td>
<td>0.01</td>
<td>0.04</td>
<td>0.01</td>
<td>0.00</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>$r$</td>
<td>0.81</td>
<td>-0.63</td>
<td>0.85</td>
<td>0.83</td>
<td>-0.74</td>
<td>-0.86</td>
<td>-0.91</td>
<td>-0.85</td>
<td>1.00</td>
</tr>
<tr>
<td>$n$</td>
<td>12</td>
<td>12</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>4</td>
</tr>
</tbody>
</table>
3.1.1.4 TURBIDITY

Turbidity varied between all sites, Waitoru Falls recording the highest reading during summer, and Ocean View recording a similarly high reading during winter (Figure 3-4). This high turbidity reading at Waitoru Falls is consistent with the regularly observed sediment layer that had been deposited on boulders at the bottom of pools at this site, an observation not commonly seen at any of the other study sites. This sediment deposition could also have been the result of very low stream discharge at this site, causing slow water velocity in some of the pools, allowing suspended sediment more time to settle out of the water column.

Turbidity readings during the winter had a mean of 8.0 NTU compared to a mean of 4.7 NTU during summer. However, there was no significant difference between summer and winter turbidity levels ($n=4$, $p=0.06$) with the $p$-value being just outside a 95% significance level. It is likely that the turbidity readings observed at each site are the combined result of stream channel damage, catchment soil type and geology, and rainfall volumes that have occurred within each catchment in the days or weeks preceding the measurement. A significant negative relationship also occurred between turbidity and pH ($n=12$, $r=-0.62$, $p=0.03$).

![Figure 3-4 Mean stream turbidity ($n=6$). Error bars = ±1 standard error.](image)

There were few correlations between turbidity and environmental variables in Table 3-4. However, during summer there was a relationship between increasing amounts of bank cover and increasing turbidity levels. Empirical evidence during stream measurement suggests that this could have occurred because bank cover was often measured where banks had been freshly undercut by...
previous formative discharge events. A high proportion of undercut banks could be contributing suspended sediment to stream water at these sites.

<table>
<thead>
<tr>
<th>p</th>
<th>Bank Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.03</td>
<td>0.00</td>
</tr>
</tbody>
</table>

Table 3-4 Pearson correlations between turbidity and other stream environmental variables. \( n=12 \) included combined summer and winter, \( n=8 \) are summer sites.

### 3.1.1.5 TEMPERATURE

Annual and diel changes in stream temperature occur at each site over time due to fluctuating air temperatures, stream shading and topographic orientation of the stream to the sun (Allan, 1995). Groundwater inputs also help maintain cool water temperatures during the summer (Allan and Castillo, 2007). While an effort was made to sample water temperatures within the closest possible time of each other on the same day, it is acknowledged that stream temperatures may have increased as readings were progressively taken throughout the day between sites sampled during summer or winter.

During summer sampling, Opanuku Tributary Stream recorded the lowest water temperature at 11°C, while Waitoru Falls Stream recorded the highest at 13.5°C (Figure 3-5). Stream temperature was between 11°C and 13°C for all summer sites and between 12.5°C and 13.5°C for all winter sites. Mean winter water temperatures were 12.9°C, higher than their summer counterparts with a mean of 11.9°C, showing a significant difference between summer and winter sites \( (n=4, \ p=0.03) \). This is thought to be due to warmer weather conditions that occurred in the days and hours preceding when winter water quality readings were taken. It could also be possible that different dissolved oxygen probes were used to record the temperature between summer and winter sampling periods, giving consistently different temperature results (although they were the same model).
A significant correlation occurred between increasing temperatures and decreasing dissolved oxygen levels.

<table>
<thead>
<tr>
<th></th>
<th>Dissolved Oxygen</th>
<th>Dissolved Oxygen</th>
<th>Conductivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>( p )</td>
<td>0.04</td>
<td>0.03</td>
<td>0.01</td>
</tr>
<tr>
<td>( r )</td>
<td>-0.60</td>
<td>-0.77</td>
<td>1.00</td>
</tr>
<tr>
<td>( n )</td>
<td>12</td>
<td>8</td>
<td>4</td>
</tr>
</tbody>
</table>

Table 3-5 Pearson correlations between stream temperature and other stream environmental variables. \( n=12 \) included combined summer and winter sites, \( n=8 \) are summer sites and \( n=4 \) are winter sites.

### 3.1.2 STREAM HABITAT

#### 3.1.2.1 DISCHARGE

Within each season, calculated stream discharges were consistent with observed stream discharges the between sites (Figure 3-6). Karekare and Opanuku recorded the highest discharges during the summer from the eight sites, while Opal Pool had the highest discharge from the four sites sampled during winter. Waitoru falls had the lowest amount of flow of all the sites, being only a fraction of Karekare and Opanuku Stream discharges. The mean discharge between the four winter sites was 0.117 m\(^3\)/s, showing significantly higher flow rates than the mean of those same four sites during summer, at 0.028 m\(^3\)/s (\( n=4, p=0.03 \)).
As stream discharges increased stream width increased also (Table 3-6), this is consistent with observations of the River Continuum Concept (Vannote et al., 1980). Sites with small stream discharges also had typically higher pH levels and increased amounts of vegetation cover measured by the densiometer.

![Graph showing mean stream discharge of all sites (n=3). Error bars = ±1 standard error.](image)

**Figure 3-6** Mean stream Discharge of all sites (n=3). Error bars = ±1 standard error.

<table>
<thead>
<tr>
<th>Study Sites</th>
<th>Discharge (m³/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Opunuku</td>
<td>0.05</td>
</tr>
<tr>
<td>Swanson</td>
<td>0.07</td>
</tr>
<tr>
<td>Waitou Falls</td>
<td>0.10</td>
</tr>
<tr>
<td>karekare</td>
<td>0.15</td>
</tr>
<tr>
<td>Opunuku Trib</td>
<td>0.20</td>
</tr>
<tr>
<td>Ocean View</td>
<td>0.25</td>
</tr>
<tr>
<td>Opal Pool</td>
<td>0.30</td>
</tr>
<tr>
<td>Whakatere</td>
<td>0.35</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Study Sites</th>
<th>Discharge (m³/s)</th>
</tr>
</thead>
<tbody>
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<td>0.05</td>
</tr>
<tr>
<td>Swanson</td>
<td>0.07</td>
</tr>
<tr>
<td>Waitou Falls</td>
<td>0.10</td>
</tr>
<tr>
<td>karekare</td>
<td>0.15</td>
</tr>
<tr>
<td>Opunuku Trib</td>
<td>0.20</td>
</tr>
<tr>
<td>Ocean View</td>
<td>0.25</td>
</tr>
<tr>
<td>Opal Pool</td>
<td>0.30</td>
</tr>
<tr>
<td>Whakatere</td>
<td>0.35</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Study Sites</th>
<th>Discharge (m³/s)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Opanuku</td>
<td>0.05</td>
</tr>
<tr>
<td>Swanson</td>
<td>0.07</td>
</tr>
<tr>
<td>Waitou Falls</td>
<td>0.10</td>
</tr>
<tr>
<td>karekare</td>
<td>0.15</td>
</tr>
<tr>
<td>Opunuku Trib</td>
<td>0.20</td>
</tr>
<tr>
<td>Ocean View</td>
<td>0.25</td>
</tr>
<tr>
<td>Opal Pool</td>
<td>0.30</td>
</tr>
<tr>
<td>Whakatere</td>
<td>0.35</td>
</tr>
</tbody>
</table>

**Table 3-6** Pearson correlations between stream discharge and other stream environmental variables. n=12 included combined summer and winter sites, n=8 are summer sites.

<table>
<thead>
<tr>
<th>p</th>
<th>pH</th>
<th>Stream Width</th>
<th>Densiometer</th>
</tr>
</thead>
<tbody>
<tr>
<td>r</td>
<td>-0.62</td>
<td>0.96</td>
<td>-0.91</td>
</tr>
<tr>
<td>n</td>
<td>12</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>
3.1.2.2 CHANNEL WIDTH

Karekare and Opanuku Stream had wider active stream channels than the rest of the sites (Figure 3-7), they also had the highest discharges during summer sampling (Figure 3-6) and this is evident in the strong correlation between stream width and stream discharge during summer (Table 3-7). Stream width also showed a significant negative relationship with densiometer vegetation cover.

![Graph showing mean stream width for each study reach (n=20 to 27). Error bars = ±1 standard error.](image)

As channel width increased, so did stream discharge, with densiometer cover measurements decreasing with channel width (Table 3-7).

<table>
<thead>
<tr>
<th></th>
<th>Discharge</th>
<th>Densiometer</th>
</tr>
</thead>
<tbody>
<tr>
<td>p</td>
<td>0.00</td>
<td>0.01</td>
</tr>
<tr>
<td>r</td>
<td>0.96</td>
<td>-0.86</td>
</tr>
<tr>
<td>n</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>

Table 3-7 Pearson correlations between active channel width and other stream environmental variables. n=8 are the summer sites.

3.1.2.3 DISTANCE FROM THE OCEAN

Study sites assumed a bi-modal distribution in distances from the ocean, with four sites within two kilometres of the ocean and four sites more than 7 km from the ocean (Figure 3-8). Physiographic differences between each of the four sites are probably caused by differing environmental conditions that result from a sites proximity to the ocean.
Figure 3-8  Study reach distance from the ocean along the stream channel.

The closer sites were to the ocean, the higher conductivity levels were (Table 3-8). This is likely to be the result of airborne salts from the ocean being flushed into streams at these sites. This was especially evident during summer, when less rain occurs to wash salts from vegetation and ground surfaces into streams, resulting in higher concentrations of salts in solution.

<table>
<thead>
<tr>
<th></th>
<th>Debris Dams</th>
<th>pH</th>
<th>Conductivity</th>
<th>Dissolved Oxygen</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>p</strong></td>
<td>0.05</td>
<td>0.01</td>
<td>0.01</td>
<td>0.02</td>
</tr>
<tr>
<td><strong>r</strong></td>
<td>-0.70</td>
<td>-0.87</td>
<td>-0.85</td>
<td>0.79</td>
</tr>
<tr>
<td><strong>n</strong></td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>

Table 3-8  Pearson correlations between distance from the ocean and other stream environmental variables.  *n*=8 are the summer sites.

### 3.1.2.4 BANK SCARRING

Bank scarring was measured as being highest during summer and winter at Opanuku Tributary stream (Figure 3-9). The least amount of bank scarring during summer was evident at Opal Pool, with only slightly more occurring at Opanuku, Waitoru Falls, Ocean View and Whakatai Streams. All four winter sites showed an increased degree of bank scarring compared to their summer equivalents, Opal Pool in particular showed a large increase. The mean length of bank scarring measured from all winter sites was 51 m, compared to 23.1 m across those four sites in summer. However, there was no significant difference between the four summer and winter sampling means at those streams (*n*=4, *p*=0.11). Turbidity was higher at three of the four winter sites (Figure 3-4), possibly showing a contribution of suspended particles to water from bank scars.
An increase in the degree of bank scarring at sites was often accompanied with an increase in the amount of vegetation cover, represented by DIFN cover at stream and bank level (Table 3-9). This could possibly be due to increased amounts of vegetation surrounding the stream binding stream banks with root mass, reducing the amount of erosion (Harding et al., 2004; Wynn et al., 2004; Niyogi et al., 2007), although root density along stream banks was not directly measured.

<table>
<thead>
<tr>
<th></th>
<th>pH</th>
<th>Stream DIFN</th>
<th>Bank DIFN</th>
</tr>
</thead>
<tbody>
<tr>
<td>p</td>
<td>0.02</td>
<td>0.02</td>
<td>0.01</td>
</tr>
<tr>
<td>r</td>
<td>-0.66</td>
<td>0.98</td>
<td>1.00</td>
</tr>
<tr>
<td>n</td>
<td>12</td>
<td>4</td>
<td>4</td>
</tr>
</tbody>
</table>

Table 3-9 Pearson correlations between bank scarring and other stream environmental variables. *n*=12 included combined summer and winter sites, *n*=4 are winter sites.

### 3.1.2.5 DEBRIS DAMS

From the eight sites sampled during summer, Opanuku, Karekare and Opanuku Tributary Streams all had the least amount of debris dams, while the greatest amount of debris dams were recorded in Whakataki Stream (Figure 3-10). Fewer debris dams were measured at three of the four winter sites compared to those four sites during summer, with Opanuku Tributary showing little change in the amount of debris dams present between seasons. The mean measure of debris dams during winter was 32.3 m, less than the mean of 45.2 m observed from those same four sites during summer, however there was no significant difference between summer and winter (*n*=4, *p*=0.343).
structurally complex debris dams were noted at Opal Pool, two at Ocean View and one at Whakatai Stream and may provide various key ecological benefits to streams that contain debris dams (Rolauffs et al., 2001; Rowe and Smith, 2003).

![Figure 3-10 Measure of debris dams partially submerged along each 150m study reach. Error bars = ±1 standard error.](image)

During the summer, sites that had the largest amount of debris dams tended to be closer to the ocean (Table 3-10). This could be indicative of particular coastal forest vegetation species that exist in these environments that contributed proportionately more woody debris to those particular streams.

Table 3-10 Pearson correlations between debris dams and other stream environmental variables. $n=12$ combined summer and winter sites, $n=8$ are summer sites.

<table>
<thead>
<tr>
<th></th>
<th>Conductivity</th>
<th>pH</th>
<th>Conductivity</th>
<th>DIFN Stream</th>
<th>DIFN Bank</th>
<th>Distance to the Ocean</th>
</tr>
</thead>
<tbody>
<tr>
<td>$p$</td>
<td>0.00</td>
<td>0.02</td>
<td>0.01</td>
<td>0.01</td>
<td>0.01</td>
<td>0.54</td>
</tr>
<tr>
<td>$r$</td>
<td>0.81</td>
<td>0.79</td>
<td>0.85</td>
<td>-0.83</td>
<td>-0.83</td>
<td>-0.70</td>
</tr>
<tr>
<td>$n$</td>
<td>12</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>
3.1.2.6 BANK COVER

Waitoru Falls Stream had the most submerged bank cover of all sites sampled during summer, with the least amount occurring at Whakatai and Ocean View Streams (Figure 3-11). All four sites sampled during winter exhibited more bank cover with a mean of 50.5 m, compared to their summer compliments that had a mean length of 33.6 m, however there was no significant difference between summer and winter ($n=4, p=0.34$).

![Graph showing meters of bank cover over 300m study reach for different study sites.](image)

Figure 3-11 Measure of overhanging and undercut bank cover submerged underwater as residential habitat for fish. Error bars = ±1 standard error. Note: True left bank =150m; True right bank =150m.

A significant relationship was present between high amounts of bank cover at sites that had waters that were more turbid (Table 3-11).

<table>
<thead>
<tr>
<th>Turbidity</th>
<th>$p$</th>
<th>$r$</th>
<th>$n$</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.00</td>
<td>0.94</td>
<td>8</td>
</tr>
</tbody>
</table>

Table 3-11 Pearson correlations between bank cover and other stream environmental variables. $n=8$ are summer sites.
3.1.3 STREAM VEGETATION COVER

Vegetation cover analysis utilised three different measurement techniques; spherical densiometer, plant canopy analyser and hemispherical photography.

3.1.3.1 DENSIOmeter

Densiometer measurements between stream centre and bank measurements were largely different at Opanuku, Karekare and Opal Pool streams (Figure 3-12). These streams also had the highest discharges and the widest stream widths (Figure 3-6 and Figure 3-7). Stream banks showed a higher degree of vegetation cover than did measurements taken at the stream centre.

![Figure 3-12 Percent vegetation cover along the stream banks and stream centre measured with the modified spherical densitometer (n=20-27). Error bars = ±1 standard error.](image)

Opanuku, Karekare and Opal Pool streams had the least amount of vegetation cover from combined bank and stream centre densiometer measurements (84.6, 86.3, 87.6% respectively), while Swanson stream had the highest amount of combined bank and stream centre cover (98%; Figure 3-13). Combined bank and stream densiometer measurements showed a highly significant negative correlation with stream discharge during summer (n=8, r=-0.91, p=0.00) and stream width during summer (n=8, r=-0.86, p=0.01). No significant correlations occurred between densiometer cover and winter variables.
Figure 3-13  Percent vegetation cover measured with the modified spherical densiometer ($n=20-27$). Measurement derived from combined bank and stream vegetation cover measurements in Figure 3-12. Error bars = ±1 standard error.

A higher amount of vegetation cover was measured at smaller streams, stream sizes indicated by smaller stream widths and smaller discharges (Table 3-12). This is consistent with the understanding that riparian vegetation will have less influence over the stream as the stream becomes larger and wider (Vannote et al., 1980).

Table 3-12 Pearson correlations between densiometer readings and other stream environmental variables. $n=8$ are summer sites.

<table>
<thead>
<tr>
<th></th>
<th>Discharge</th>
<th>Stream Width</th>
</tr>
</thead>
<tbody>
<tr>
<td>$p$</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>$r$</td>
<td>-0.91</td>
<td>-0.86</td>
</tr>
<tr>
<td>$n$</td>
<td>8</td>
<td>8</td>
</tr>
</tbody>
</table>

3.1.3.2 PLANT CANOPY ANALYSER

Stream cover (DIFN) measured at both bank and stream levels was consistently between 95-99% at most sites, except Ocean View, Opal Pool and Whakatai Streams (Figure 3-14). These latter three streams exhibited less stream level cover than the others at 81, 91 and 64% respectively. These three sites were selected a-priori for winter sampling round, as they represented a cross-section of cover categories. Opanuku Tributary was also chosen a-priori for winter sampling as a site with a high degree of DIFN cover (99.1%).
Chapter 3 | The relationship between stream cover and stream insect activity

Many relationships existed between DIFN cover and stream environmental variables, with strong positive correlations between DIFN cover measurements taken at both bank (Table 3-13) and stream (Table 3-14) levels. This is to be expected as the two measurements are auto-correlated. Sites with less DIFN cover also exhibited higher conductivity levels. This could be because those sites were located close to the coast, in which the coastal vegetation type exhibited a more spatially open forest interior environment.

Table 3-13 Pearson correlations between DIFN cover at stream height and other stream environmental variables. n=8 are summer sites and n=4 are winter sites.

<table>
<thead>
<tr>
<th></th>
<th>Debris Dams</th>
<th>pH</th>
<th>Conductivity</th>
<th>Bank DIFN</th>
<th>Bank Scarring</th>
<th>Bank DIFN</th>
</tr>
</thead>
<tbody>
<tr>
<td>p</td>
<td>0.01</td>
<td>0.02</td>
<td>0.01</td>
<td>0.00</td>
<td>0.03</td>
<td>0.03</td>
</tr>
<tr>
<td>r</td>
<td>-0.83</td>
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<td>-0.86</td>
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<td>0.98</td>
<td>0.98</td>
</tr>
<tr>
<td>n</td>
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<td>8</td>
<td>4</td>
<td>4</td>
</tr>
</tbody>
</table>

Figure 3-14 Percent stream cover measured with the LI-COR Plant Canopy Analyser. DIFN measurements are from stream level means and bank-full height means. Error bars = ±1 standard error.
Table 3-14 Pearson correlations between DIFN cover at bank height and other environmental variables. \( n=8 \) are summer sites and \( n=4 \) are winter sites.

<table>
<thead>
<tr>
<th></th>
<th>Debris Dams</th>
<th>pH</th>
<th>Conductivity</th>
<th>Stream DIFN</th>
<th>Bank Scarring</th>
<th>Bank DIFN</th>
</tr>
</thead>
<tbody>
<tr>
<td>( p )</td>
<td>0.01</td>
<td>0.02</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.03</td>
</tr>
<tr>
<td>( r )</td>
<td>-0.83</td>
<td>-0.78</td>
<td>-0.91</td>
<td>0.98</td>
<td>1.00</td>
<td>0.98</td>
</tr>
<tr>
<td>( n )</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>8</td>
<td>4</td>
<td>4</td>
</tr>
</tbody>
</table>

3.1.3.3 HEMISPHERICAL PHOTOGRAPHY

Using the method of photo capture and photo processing employed in this study, with light thresholds set at 15 and 215 (see section 0), Figure 3-15 illustrates how vegetation cover over the stream was distributed at each stream reach. The amount of sky fraction visible over each stream is similar, with Opanuku stream showing the largest proportion of open sky (7.2%) and Waitoru Falls showing the smallest proportion of open sky (2.6%).

The amount of far cover was approximately reciprocal to the amount of near cover utilising this hemispherical photography technique. The proportion of near cover is the most favourable measurement to use for further analysis, because any increase in the measurement of near cover will automatically mask out the proportion of far cover visible behind. The largest proportion of near cover was recorded at Waitoru Stream (58.9%) and the least amount of near cover was recorded at Karekare stream (38.9%). The comparison in the amount of near cover between these two measurements was also observed in the field.

The only correlation relationship that occurred was that between the amount of near cover and the amount of far cover during summer \( (n=8, p=-0.00, r=0.98) \) and during winter \( (n=4, p=0.01, r=-0.99) \). Note should be made to the lack of a significant relationship between DIFN cover and percent sky in this method, it is likely that this indicates a lack of accuracy from the hemispherical photography method.
3.2 DRIFT ACTIVITY RESULTS

While an effort was made to ensure samplers were not sampling during or immediately after prolonged periods of rain, it was inevitable that water levels did fluctuate during sampling, due intermittent periods of rain. Although there was always a flow of water going through each sampler and drift contents were always obtained, the results must be interpreted with this minor water level fluctuation in mind.

Items that drifted into the sampler are assumed to have stayed in the sampler; however this may not have always been the case. On one occasion, a beetle was observed crawling out of the sampler, (albeit with some difficulty) despite the smooth plastic entrance to the sampler providing little traction and another occasion saw a weta intentionally crawling underwater on the surface of boulder. On another occasion, a small Dipteran fly was observed flying out of the sampler. The effect of this ‘insect escape’ error on drift results is assumed to be minimal. It is also possible that some small banded kokopu and bullies that were occasionally captured in the drift samplers were feeding on insects inside the sampler. This error is assumed to be minimal also, given the small size of the sampler and the difficulty a fish would have when attempting to feed inside them.
3.2.1 Aquatic and Terrestrial Composition

The highest densities of invertebrates captured in drift occurred at Ocean View Stream (4.39 \( \text{m}^3 \); Figure 3-16). Opal Pool and Swanson also exhibited high densities of invertebrates compared to the other sites (2.98 and 2.73 \( \text{m}^3 \) respectively), while the least amount of insect drift occurred at Opanuku Trib WINTER, Opanuku, Karekare and Opal Pool WINTER streams (0.43, 0.52, 0.23 and 0.51 \( \text{m}^3 \) respectively).

Larger proportions of terrestrial invertebrates were captured relative to aquatic invertebrates at Ocean View (64% terrestrial), Opal Pool (84% terrestrial), Opanuku Tributary (76% terrestrial) and Swanson streams (81% terrestrial). Noticeably more aquatic invertebrates were captured relative to terrestrial invertebrates at Whakatia Stream during summer (18% terrestrial). At all other sites, terrestrial and aquatic insects occurred in approximately equal proportions. There were significant differences in the amount of insects captured between each summer site (\( n=8, p=0.00 \)) and between each winter site (\( n=4, p=0.00 \)). More specifically, there were significantly more terrestrial insects captured during summer than there were during winter (\( n=24, p=0.00 \)) at the four sites.

![Aquatic and terrestrial mean drift composition of insects captured in drift per m³ of water discharged at each site in a discontinuous 72 hour period (42 hours of day and 30 hours of night per bar; \( n=6 \)]. Error bars = ±1 standard error.

At sites where there was a high degree of bank DIFN cover, there were lower quantities of all insects in drift (Table 3-15). At sites that had high amounts of aquatic insects relative to sites with lower amounts of aquatic insects, there were more debris dams and higher conductivity. These sites also
had less DIFN cover measured at bank height. At sites with relatively high amounts of terrestrial insects, there was less bank DIFN cover, less bank scarring and less bank cover.

Table 3-15 Pearson correlations between environmental variables and a) all insects, b) aquatic insects and c) terrestrial insects. \( n=12 \) combined summer and winter sites, \( n=8 \) are summer sites and \( n=4 \) are winter sites

<table>
<thead>
<tr>
<th></th>
<th>a)</th>
<th>b)</th>
<th>c)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bank DIFN</td>
<td>Debris Dams</td>
<td>Conductivity</td>
</tr>
<tr>
<td>( p )</td>
<td>0.04</td>
<td>0.01</td>
<td>0.01</td>
</tr>
<tr>
<td>( r )</td>
<td>-0.96</td>
<td>0.69</td>
<td>0.73</td>
</tr>
<tr>
<td>( n )</td>
<td>4</td>
<td>12</td>
<td>12</td>
</tr>
</tbody>
</table>

### 3.2.2 NET TERRESTRIAL DRIFT

The net abundance values of terrestrial insects in whole-stream drift are shown in Table 3-16. The most insects flowing down the stream at any one time (net drift) occurred at Opal Pool during summer, while Waitorū Falls exhibited the lowest net amount of insects. When comparing net drift between the seasons, only one site out of the four (Whakatāi) showed increased net transport of insects during winter. This site was also the only one that showed an increase in insect densities during winter sampling.

Table 3-16 Net movement and density of terrestrial insects in drift at each site. Net movement is quantified as the number of terrestrial insects per minute of whole stream discharge. Density is quantified as the number of terrestrial insects per m\(^3\) of stream water.

<table>
<thead>
<tr>
<th></th>
<th>Opanuku</th>
<th>Swanson</th>
<th>Waitorū Falls</th>
<th>Karēkāre</th>
<th>Opanuku Trib</th>
<th>Ocean View</th>
<th>Opal Pool</th>
<th>Whakatāi</th>
</tr>
</thead>
<tbody>
<tr>
<td>Summer</td>
<td>Terrestrial Drift Insects/minute</td>
<td>1.8</td>
<td>3.8</td>
<td>0.1</td>
<td>2.6</td>
<td>2.3</td>
<td>4.8</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>Terrestrial Drift Insects/m(^3)</td>
<td>0.20</td>
<td>2.21</td>
<td>0.61</td>
<td>0.22</td>
<td>1.28</td>
<td>2.82</td>
<td>2.49</td>
</tr>
<tr>
<td>Winter</td>
<td>Terrestrial Drift Insects/minute</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>1.5</td>
</tr>
<tr>
<td></td>
<td>Terrestrial Drift Insects/m(^3)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.15</td>
</tr>
</tbody>
</table>
3.2.3 DAY AND NIGHT DRIFT COMPOSITION

The most dissimilarity between day and night aquatic drift activities occurred at Whakatai and Waitoru Streams, where more aquatic drift occurred during night (Figure 3-17). The largest difference in terrestrial drift activity between day and night periods occurred at Swanson and Opanuku Tributary Streams, where more terrestrial drift occurred during the night relative to daytime. The Swanson night result exhibits a large error bar, owing primarily to one night drift sample where large amounts of terrestrial Arthropoda Collembola (springtail) were present in all three sub-samplers. This resulted in noticeably larger amounts of insects in drift in that one particular night sample.

During the summer, Opal Pool had a large proportion of terrestrial insects (Figure 3-18), largely the result of Dipteran leaf miner larvae present in proportionally high numbers in shed kanuka tree leaves. The high proportion of aquatic drift present at Whakatai during summer was caused by a low amount terrestrial drift and increased amounts of Ephemopteran larvae, Trichopteran larvae and Potamopyrgus snails. There was no significant difference between the amount of terrestrial or aquatic insects that occurred in drift during summer (n=48, p=0.17) or during winter (n=24, p=0.28).

Figure 3-17  Day and Night mean drift composition per m$^3$ stream discharge (n=3). Error bars = ±1 standard error.
Percentage composition of terrestrial and aquatic drift components per cubic metre of water showed higher proportions of aquatic drift occurring during the night relative to day at all sites, with the exception of Opal Pool during the winter (Figure 3-18). There was no difference in the amount of terrestrial insects occurring in drift between day and night in summer \((n=24, p=0.41)\) or winter \((n=12, p=0.32)\). There were significantly more aquatic insects occurring in night drift than there was in day drift during summer \((n=24, p=0.00)\) and winter \((n=12, p=0.03)\).

**Figure 3-18** Day and Night drift proportions of invertebrates per m\(^3\) of stream discharge \((n=3)\).
3.2.4 NIGHT DRIFT ACTIVITY

Figure 3-19 illustrates the relative composition of terrestrial and aquatic insects in drift during night. In night drift samples, there was no difference in the amount of insects that occurred in drift during summer ($n=24, p=0.51$) or winter ($n=12, p=0.89$). There was however, significantly higher densities of terrestrial insects in drift during summer compared to winter ($n=12, p=0.00$). There were also significantly higher densities of aquatic insects during the night in summer compared to winter ($n=12, p=0.00$).

![Figure 3-19 Night mean invertebrate drift composition ($n=3$). Error bars = ±1 standard error.](image)

A number of significant relationships were present between the amount of insects captured in drift at night and environmental variables (Table 3-17). The amount of insect drift occurring at each site was related to the amount of drift that occurred during the day. During winter, sites with the most bank DIFN cover had the least amount of total (Table 3-17a), terrestrial (Table 3-17b), and aquatic (Table 3-17c) insects in drift.
Table 3-17 Pearson correlations between environmental variables, other insect quantities and a) all insects in drift at night, b) terrestrial insects in drift at night and c) aquatic insects in drift at night. n=12 are summer and winter sites, n=8 are summer sites and n=4 are winter sites

<table>
<thead>
<tr>
<th></th>
<th>a) Day Drift Insects</th>
<th>Day Discharge</th>
<th>Discharge</th>
<th>Stream Width</th>
<th>Densiometer</th>
<th>Bank Cover</th>
<th>Bank Scarring</th>
<th>Bank DIFN</th>
</tr>
</thead>
<tbody>
<tr>
<td>p</td>
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<td>0.03</td>
<td>0.02</td>
<td>0.02</td>
<td>0.02</td>
<td>0.04</td>
<td>0.02</td>
<td>0.02</td>
</tr>
<tr>
<td>r</td>
<td>0.67</td>
<td>-0.64</td>
<td>-0.78</td>
<td>-0.81</td>
<td>0.80</td>
<td>-0.96</td>
<td>-0.98</td>
<td>-0.98</td>
</tr>
<tr>
<td>n</td>
<td>12</td>
<td>12</td>
<td>8</td>
<td>8</td>
<td>8</td>
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<td>4</td>
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<table>
<thead>
<tr>
<th></th>
<th>b) Day Drift Insects</th>
<th>Day Terrestrial Drift Insects</th>
<th>Night Aquatic Drift Insect</th>
<th>Aquatic Drift Insects</th>
<th>Day Terrestrial Drift Insects</th>
<th>Bank Scarring</th>
<th>Stream DIFN</th>
<th>Bank DIFN</th>
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</thead>
<tbody>
<tr>
<td>p</td>
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<table>
<thead>
<tr>
<th></th>
<th>c) Day Aquatic Drift Insects</th>
<th>Debris Dams</th>
<th>Conductivity</th>
<th>Discharge</th>
<th>Debris Dams</th>
<th>Conductivity</th>
<th>Bank DIFN</th>
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<th>Bank Scarring</th>
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<tbody>
<tr>
<td>p</td>
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<td>0.01</td>
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<td>0.02</td>
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<td>4</td>
<td>4</td>
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</tbody>
</table>
3.2.5 DAY DRIFT ACTIVITY

Similar drift patterns were evident during the day when comparing seasons (Figure 3-20), showing lower amounts of aquatic drift during winter compared to those same four sites sampled during summer. In day drift samples, there was no difference in the amount of insects that occurred in drift during summer \((n=24, p=0.17)\) or winter \((n=12, p=0.16)\). There was no significant difference between terrestrial insect densities during the day in summer compared to winter \((n=12, p=0.07)\). There were significantly higher densities of aquatic insects in drift during the day in summer compared to winter \((n=12, p=0.01)\).

During the day, at sites that had a high proportion of bank DIFN cover, there were fewer terrestrial insects in drift (Table 3-18b) but higher amounts of aquatic insects in drift (Table 3-18c).
Table 3-18  Pearson correlations between environmental variables and other insect quantities, and a) all insects in drift during day, b) terrestrial insects in drift during day and c) aquatic insects in drift during day. $n=12$ summer and winter sites, $n=8$ are summer sites and $n=4$ are winter sites.

<table>
<thead>
<tr>
<th></th>
<th>Discharge</th>
<th>Discharge</th>
<th>Stream Width</th>
<th>Densiometer</th>
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<th>Scarring</th>
<th>Bank DIFN</th>
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<tr>
<td><strong>a)</strong></td>
<td></td>
<td></td>
<td></td>
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<td></td>
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<tr>
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<th>Night Drift Insects</th>
<th>Aquatic Drift Insects</th>
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<td></td>
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<tr>
<td>$p$</td>
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<th>Day Terrestrial Drift Insects</th>
<th>Terrestrial Drift Insects</th>
<th>Conductivity</th>
<th>Day Terrestrial Drift Insects</th>
<th>Conductivity</th>
<th>Bank DIFN</th>
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3.2.6 TERRESTRIAL INSECTS AND DETRITUS IN DRIFT

A significant, but weak linear relationship existed between the amount of dried small leaf and twig detritus captured in each drift sample and the number of insects caught in that sample (Figure 3-21). This shows that in general, as the amount of small leaf litter captured in drift increases, so will the amount of insects captured in that drift sample. Ten drift samples contained sand that had become entrained in drift; these added extra weight to each dried drift detritus sample and therefore were excluded from the graph presented in Figure 3-21.

There were no significant Pearson correlations between the amount of drift detritus and environmental variables at each site.

![Figure 3-21 Terrestrial insect numbers and dry drift detritus weights per drift sampler (n=216) after a 14 hour sampling day or a 10 hour sampling night. Line of best fit shows linear regression (r=0.32, p=0.00). Ten samples that contained sand were omitted. Note log scale.](image)

3.3 STREAM DISCHARGE/WIDTH AND INSECT DENSITY RESULTS

A highly significant positive correlation existed between stream width and stream discharge at each of the sites during summer ambient flow levels (Figure 3-22a). However, during elevated winter flow rates this linear relationship was weak and not as clearly defined (Figure 3-22b). The significant correlation is consistent with the general geomorphic properties we would expect to observe in a stream along the river continuum gradient (Vannote et al., 1980).
Figure 3-22  Relationship between stream widths and stream discharges during a) summer \((n=8, p=0.00**)\) and during b) winter \((n=4, p=0.52)\). Confidence bands = ±1 standard error of the regression line. **= 99% chance that the correlation is significant.

A non-significant negative relationship was evident between stream widths and insect drift densities during summer (Figure 3-23a). This relationship was not evident during winter, with the regression line in Figure 3-23b exhibiting a weak relationship with a non-significant correlation value.

Figure 3-23  Linear regression relationship between stream widths and drift densities for a) summer \((n=8, p=0.10)\) and b) winter \((n=4, p=0.73)\). Confidence bands = ±1 standard error of the regression line.
An evident, but non-significant negative relationship between stream discharge and insect drift densities at all sites for both seasons is illustrated in Figure 3-24, illustrating a decreasing amount of insects in drift as stream discharges increase.

Figure 3-24 Stream discharges and drift densities for all sites during both summer and winter seasons ($n=12$, $p=0.09$). Confidence bands = ±1 standard error of the regression line.

3.4 STREAM COVER AND INSECTS IN DRIFT RESULTS

3.4.1 SEASONAL VARIATION IN INSECT DRIFT

The relationship between DIFN cover over the stream and insects captured in drift for each season is illustrated in Figure 3-25a, showing an imperceptible regression slope and a non-significant negative correlation value during summertime. Figure 3-25b illustrates the same negative regression trend, but with correlation values that are highly significant.

Figure 3-25 DIFN cover (taken at bank height) over the streams and insects captured in drift during a) summer ($n=8$, $p=0.48$) and b) winter ($n=4$, $p=0.39$). Confidence bands = ±1 standard error of the regression line.
3.4.2 SEASONAL VARIATION IN TERRESTRIAL AND AQUATIC DRIFT

The relationship between DIFN cover and aquatic or terrestrial insects in drift for both summer and winter seasons is presented in Figure 3-26. A weak relationship is visible between increasing DIFN cover with increasing amounts of terrestrial insects captured in drift during summer (Figure 3-26a), however the inverse is evident during winter where a significant negative relationship exists showing terrestrial insect activity to increase with lower amounts of stream cover (Figure 3-26b).

The opposite relationship trends tended to occur for the aquatic proportion of stream drift. This significant relationship is shown in Figure 3-26c where aquatic insects decrease with increasing stream cover in summer. The opposite occurred during winter where the higher numbers of aquatic insects were present in stream drift as stream cover increased (Figure 3-26d); however the significance of this result is weak.
3.4.3 **DIRUNAL VARIATION IN DRIFT BY SEASON**

The relationship between DIFN cover over the stream and the amount of insects captured in drift during the day and night time is illustrated in Figure 3-27. The relationship between DIFN cover and the amount of insects in drift during the day in summer shows a weak negative non-significant relationship (Figure 3-27a), while the opposite occurs during the day in winter-time, also in a non-significant relationship (Figure 3-27b).

Night time insect activity showed very little correlation with DIFN cover during the summer (Figure 3-27c), however a significant relationship occurred in the winter with increasing DIFN cover and decreasing insect abundance occurring at night-time (Figure 3-27d).

**Figure 3-26**  DIFN cover (at bank height) over streams and terrestrial insects in drift during a) summer ($n=8$, $p=0.26$) and b) winter ($n=4$, $p=0.03^*$). DIFN cover (at bank height) over streams and aquatic insects in drift during c) summer ($n=8$, $p=0.04^*$) and d) winter ($n=4$, $p=0.06$). Confidence bands = ±1 standard error of the regression line. $^*$= A 95% chance that the correlation relationship is significant.
Figure 3.27  Relationships between DIFN cover (at bank height) over the stream and drift insects during a) day-time in summer ($n=8$, $p=0.44$), b) day-time in winter ($n=4$, $p=0.11$), c) night-time in summer ($n=8$, $p=0.79$) and d) night-time in winter ($n=4$, $p=0.02$). Confidence bands $= \pm 1$ standard error of the regression line. * A 95% chance that the correlation relationship is significant.

3.4.4 DIRURNAL AND SEASONAL VARIATION IN TERRESTRIAL AND AQUATIC DRIFT

Graphs relating DIFN cover to aquatic and terrestrial insects by day and night for both summer and winter season are shown in Figure 3.28. The only significant relationships that occurred during the summer were between increasing bank DIFN cover and decreasing numbers of aquatic insects in drift at night-time (Figure 3.28g). During winter, significant relationships were evident between increasing amounts of DIFN cover and decreasing amounts of terrestrial (Figure 3.28d) and aquatic insects (Figure 3.28h) by night.

Regression lines and correlation analysis between increasing stream DIFN cover and increasing terrestrial insects drift at night time during winter were the only significant drift parameters
measured when utilising the stream DIFN cover parameter as the dependant variable. All regressions and correlation measurements for stream DIFN measurements are provided in appendix Table 6-2.
The relationship between DIFN cover (at bank height) and a) terrestrial insects by day during summer ($n=8$, $p=0.58$), b) terrestrial insects by day during winter ($n=4$, $p=0.06$) c) terrestrial insects by night during summer ($n=8$, $p=0.78$), d) terrestrial insects by night during winter ($n=4$, $p=0.01^{**}$), e) aquatic insects by day during summer ($n=8$, $p=0.15$), f) aquatic insects by day during winter ($n=4$, $p=0.81$), g) aquatic insects by night during summer ($n=8$, $p=0.04^{*}$), h) aquatic insects by night during winter ($n=4$, $p=0.04^{*}$). Confidence bands $= \pm 1$ standard error of the regression line. $^{**}$= A 99% chance that the correlation relationship is significant. $^{*}$= A 95% chance that the correlation relationship is significant.
3.5 COVER PROXIMITY AND INSECTS IN DRIFT RESULTS

This section compared insect drift capture data to the amount of vegetation in close proximity to the stream surface, by way of linear regression relationship analysis.

3.5.1 SEASONAL VARIATION IN INSECT DRIFT

An insignificant amount of correlation exists between the amount of cover that is within close proximity to the stream and the amount of all insects captured from stream drift in summer (Figure 3-29a) and winter (Figure 3-29b).

![Graphs showing seasonal variation in insect drift](image)

Figure 3-29 Close vegetation cover and insects in drift during a) summer ($n=8$, $p=0.51$) and b) winter ($n=4$, $p=0.98$). Confidence bands = ±1 standard error of the regression line.

3.5.2 SEASONAL VARIATION IN TERRESTRIAL AND AQUATIC DRIFT

Figure 3-30 shows the relationships observed between the amount of near cover and the amount of invertebrates present in drift. During summer, weak correlations were observed between increasing amounts of close vegetation cover and decreasing amounts of terrestrial insects in drift (Figure 3-30a), while the opposite occurred with aquatic invertebrates during summer (Figure 3-30c). These two relationships were not significant however. Significant correlations were observed during winter with regression lines showing terrestrial insects in drift increasing as the amount of near cover increased (Figure 3-30b) and aquatic insects in drift during winter decreasing as the amount of near cover increased (Figure 3-30d).
3.5.3 DIURNAL VARIATION IN DRIFT BY SEASON

There were no significant relationships between the amount of near cover and insects in drift during day or night during summer or winter. The regression lines during summer show little relationship between the variables. Relationships are evident during winter as seen by the regression line in Figure 3-31b (decreasing insects by day as near cover increases) and Figure 3-31d (increasing insects by night as near cover increases). However, both summer and winter comparisons hold non-significant correlation values.
Figure 3-31 Graphs show the linear regression lines for the relationship between the amount of close vegetation cover and a) the amount of insects in drift at day during summer \((n=8, p=0.50)\) b) the amount of insects in drift at day during winter \((n=4, p=0.77)\) c) the amount of insects in drift during night in summer \((n=8, p=0.79)\), and d) the amount of insects in drift at night during winter \((n=4, p=0.90)\). Confidence bands = ±1 standard error of the regression line.
3.5.4 DIURNAL AND SEASONAL VARIATION IN TERRESTRIAL AND AQUATIC DRIFT

No significant relationships were shown in Figure 3-32 between the amount of near cover and terrestrial and aquatic insects in drift by day or night periods.

\[ r = -0.29 \]

\[ r = -0.04 \]

\[ r = -0.26 \]

\[ r = 0.13 \]
Figure 3-32  Linear regressions showing the relationship between the amount of close vegetation and a) terrestrial insects in drift by day during summer \((n=8, p=0.48)\), b) terrestrial insects by day during winter \((n=4, p=0.96)\), c) terrestrial insects by night during summer \((n=8, p=0.53)\), d) terrestrial insects by night during winter \((n=4, p=0.87)\), e) aquatic insects by day during summer \((n=8, p=0.62)\), f) aquatic insects by day during winter \((n=4, p=0.29)\), g) aquatic insects by night during summer \((n=8, p=0.51)\), h) aquatic insects by night during winter \((n=4, p=0.91)\). Confidence bands = ±1 standard error of the regression line.

### 3.6  DISCUSSION

#### 3.6.1  REVIEW OF THE IMPORTANCE OF INSECT SOURCES

Donations of terrestrial energy that are contributed to the stream, essentially arthropod ‘rain’ (Chan et al., 2008) from allochthonous sources is an important component in stream ecology as food for aquatic consumers (Baxter et al., 2005). Edwards and Huryn (1996) found differences in the biomass of terrestrial insects that enter streams from pasture land use, when compared to that from tussock or from native forest land uses. Romero, Gresswell et al. (2005) also observe that riparian vegetation has important influences on the provision of terrestrial invertebrates to streams. Chan,
Zhang et al. (2008) showed marked differences in terrestrial drift composition and abundance between shrubland and forest sites. Those three studies demonstrate that land use regimes in riparian zones are important predictors of terrestrial inputs to stream drift.

Explaining the specific causal mechanisms that determine terrestrial invertebrate contribution to stream drift within a forest can be a difficult task (Edwards and Huryn, 1996), considering the complex temporal (Nakano et al., 1999a) and spatial (Cloe and Garman, 1996; West et al., 2005) heterogeneity of terrestrial invertebrate densities in drift. The present study found there to be a great degree of variability between sites, seasons and diurnal cycles of terrestrial drift components (i.e., Figure 3-17), results also supported by Cloe and Garman (1996) and West, Jowett et al. (2005). It is likely that this heterogeneity in terrestrial drift density is due to multiple environmental drivers, for example; air temperature (Edwards and Huryn, 1995), riparian vegetation species type (Mason and Macdonald, 1982) or forest succession stage (Greene et al., 2008).

3.6.2 VEGETATION PROXIMITY AND INSECT CONTRIBUTIONS

When looking at the difference between native and exotic pine forests and their effect on native fish populations, Rowe, Chisnall et al. (1999) found there to be no significant difference between the two forest stream types. They discuss that:

"Most of the exotic forest sites sampled contained a dense understory of native shrubs and tree ferns similar to that occurring in native forest sites. This understory, and particularly the riparian vegetation, may be more important for fish habitat in forested streams than forest type."

Further elaborating on this statement, it may be that this understory or ‘close proximity’ vegetation not only provides morphological habitat features suitable for native fish residence, but that it also contributes proportionately higher amounts of terrestrial insects to the stream than from canopy vegetation. Therefore, fish have a dietary advantage living in streams with greater proportions of close proximity riparian vegetation compared to residence in streams with a high canopy, but little understory vegetation. In their study of terrestrial invertebrate inputs in resource budgets of salmonid fishes in forest and grassland streams, Kawaguchi and Nakano (2001) allude to overhanging vegetation possibly enhancing the input of terrestrial invertebrates that have little or no flying ability during summer.

It could be surmised that canopy vegetation provides the primary temperature regulation (Blann et al., 2002), bank stability (Wynn et al., 2004) and morphological habitat properties (such as debris dams; Rowe et al., 2002; Baker and Smith, 2007) to the stream. Understory vegetation may supply morphological, temperature and bank stability properties to the stream too, but may be of greater
importance as an insect contributor, providing proportionately more insects to the stream surface than far canopy vegetation does.

The present study used a hemispherical photography technique to estimate vegetation cover proximity, primarily because it can provide consistent and informative measures of vegetation cover estimation over streams (Englund et al., 2000; Kelley and Krueger, 2005). Exploration of the hemispherical photography technique specifically for vegetation cover proximity estimation in the present study was largely experimental. Although the method may have picked up variation in close cover between the sites, it is thought that the effects between sites may have been normalised due to the camera light compensation settings. An overestimation of close cover could also occur due to differing bank heights between sites, for example, if the technique was being used in a canyon, then the canyon walls would come out dark in the photo and would be interpreted as close cover. This study utilized a fish-eye camera lens that only provided a 167° field of view and this was insufficient for capturing a large proportion of close cover, as would occur if a true 180° fish-eye lens was used. The effect of this limited viewing angle was particularly limiting when the camera was mounted on a tripod (approximately 0.5m off the stream surface), which effectively excluded all vegetation on the stream banks below the camera and close to the water’s surface.

Future attempts at refining this technique may consider taking hemispherical canopy photos at night and illuminating vegetation from underneath. This could be done with a ring-flash that would produce light evenly around the lens to reduce any shadow effects behind leaves in the photo. Taking photographs during darkness would effectively minimize or even exclude light conditions that fluctuate during the day, even under an isotropic sky. The artificial light projected from underneath vegetation would light up close vegetation brightly and become consistently more diffuse further away from the camera. This light parameter could then be trans-located with the camera equipment between study sites, allowing light to be applied in a controlled and consistent manner.

### 3.6.3 DIURNAL AND SEASONAL EFFECTS

Temporal variation in drift densities were observed in this study, with significantly higher rates of terrestrial drift occurring during the summer, compared to drift rates observed during winter. West, Jowett et al. (2005) observed the same seasonal fluctuation in terrestrial drift densities under native forest sites in the Coromandel. Kawaguchi and Nakano (2001) and Allan, Wipfli et al. (2003) also observed increasing summer terrestrial drift under both deciduous forest and grassland. Cloe and Garman’s study (1996) showed increasing amounts of terrestrial arthropod inputs from their headwater streams during summer, also citing a number of other authors that observed the same
seasonal pattern. However, a number of these previous studies were conducted under deciduous vegetation types and this information may not be directly applicable to New Zealand streams that are typically boarded by native evergreen vegetation. A surprisingly limited amount of information could be found on diurnal and seasonal variation in terrestrial drift from under evergreen forest vegetation types.

When assessing insects numerically, as in this study; terrestrial drift activity showed no evidence of a response to diurnal cycles across both seasons. During winter however, there was a significantly higher density of terrestrial components in night drift compared to day drift, this pattern was not observed during summer. Variation between day and night densities of terrestrial drift were observed in a high country pasture stream by Edwards and Huryn (1995), where they showed the abundance of terrestrial invertebrates in drift to be higher at day than at night. Under deciduous vegetation during summer, Nakano, Kawaguchi et al. (1999a) observed the input of falling terrestrial invertebrates to a stream to peak around dusk, decrease during night and peak again towards sunrise, with a lower constant activity rate during the day. This would suggest that in their study, terrestrial invertebrates exhibited pronounced crepuscular peaks in activity. Given that drift samples in the present study were collected at sunrise and sunset, if any crepuscular peaks did occur, they would have been split between both day night samples.

3.6.4 STREAM COVER

Does an increase in riparian vegetation cover over the stream result in a corresponding increase in the amount of terrestrially derived invertebrate drift? The significance of cover relationships with terrestrial drift in the present study were varied, depending on the temporal resolution with which the invertebrate density variable was assessed. Few relationships were present when assessing vegetation cover and total invertebrate drift. However, breaking down invertebrates into terrestrial and aquatic components suggested that two shared an inverse relationship. Density of aquatic insect drift was suppressed by cover in summer and increased by cover during winter.

Terrestrial insect densities in drift increased with cover during summer, but decreased with cover during winter, represented in Figure 3-33. This result may provide evidence of a relationship when high amounts of cover provide a larger amount of habitat for terrestrial insects to take advantage of the summer season to increase their functional activity (red line in Figure 3-33). In this case, more stream cover would advantage terrestrial activity and make seasonally active insects more prone to falling into the stream. Considering that overall terrestrial insect activity is a function of increased air temperatures in summer (Mason and Macdonald, 1982; Edwards and Huryn, 1995), in the present study, higher relative proportions of terrestrial insects were observed to occur during
summer compared to winter. This proportionate difference in terrestrial activity between seasons is also represented in Figure 3-33. During winter, stream cover may act to retard the volumes of terrestrial insects that occur in stream drift.

Figure 3-33 Hypothetical relationship between the amount of terrestrial insects in stream drift and more than 50% vegetation cover over the stream.

In a study conducted by Greene et al. (2008), they conclude that the succession stage of riparian forest can also affect invertebrate availability in forest ecosystems. In their article, they refer to previous studies of forest vegetation that show above ground biomass to reach its maximum during early succession (50-70 years), with above ground biomass then decreasing as a forest reaches maturity (100-200 years). This loss of above ground biomass may also affect vegetation cover over streams as a function of forest age since last disturbance, particularly in the Waitakere Ranges, given its previous logging and land clearance history (Harvey and Harvey, 2006). If cover reduction is a function of forest age due to loss of above ground biomass in mature forests, it is possible that forest age may affect the supply of insects to streams.

The DIFN vegetation cover parameter utilised in the present study gives measurements of up to 100% vegetation cover; however this does not provide a means of quantifying vegetation density as a proxy for standing crop biomass. Utilising a light measurement parameter rather than the analogous DIFN cover parameter would allow assessment of vegetation density in the forest canopy, to a point exceeding the maximum 100% cover provided by DIFN assessment methods (to a potential maximum excluding all light). It is possible that the amount of above ground biomass could provide further habitat for invertebrates, and increase the surface area that they could fall from into the stream.

Lower temperatures and variable weather conditions may affect insects that live in above ground environments more readily compared to those living in or on the ground layer (Figure 3-34). Winter
insect activity may be more limited to ground dwelling species that live in humus and soil layers. This soil humus environment provides more stable and predictable conditions that reduce diurnal and seasonal humidity and temperature oscillations (Bruckner, 1998). This is important as temperature and humidity can affect abundance of some terrestrial insect species that reside in the soil zone (Lynch et al., 1988; Koivula et al., 1999; Kuznetsova, 2007).

Air temperature relationships with terrestrial insects have been demonstrated by Edwards and Huryn (1995) who determined that biomass contributions of winged insects entering a high country pasture stream exhibited a response to variation in air temperature. They also found the abundance

![Hypothetical illustration of insect activity during summer and winter seasons in the ground layer and in the above ground (canopy) layer.](image)

**Figure 3-34** Hypothetical illustration of insect activity during summer and winter seasons in the ground layer and in the above ground (canopy) layer.

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### 3.6.5 QUANTITY AND TIMING OF TERRESTRIAL INPUTS

Increases of stream discharge cause the wetted perimeter to laterally migrate and envelop previously un-submerged areas along the stream margin. This movement has the potential to entrain in drift any terrestrial organisms that were in close proximity to the stream’s wetted margin (O’Hop and Wallace, 1983; Edwards and Huryn, 1995). Considering this, we could expect to see a pulse of terrestrial drift as stream levels rise, followed by a subsequent reduction in terrestrial drift densities as the stream discharge ceases to increase and begins to subside. The same type of pulse could occur during the beginning of a rain, when precipitation flushes insects from vegetation and into the stream (Mason and Macdonald, 1982; O’Hop and Wallace, 1983). In the present study, a relationship existed between the amount of drift detritus and number of terrestrial insects captured in each drift sample (Figure 3-21), a relationship also observed by O’Hop and Wallace (1983). This mutual connectedness between the two factors suggest that the influence of weather activity such as wind or rain could provide a mechanism for dislodging insects and leaf detritus from overlaying vegetation and stream banks, subsequently depositing them in stream drift.

Air temperature relationships with terrestrial insects have been demonstrated by Edwards and Huryn (1995) who determined that biomass contributions of winged insects entering a high country pasture stream exhibited a response to variation in air temperature. They also found the abundance
of wingless insects was not so much related to temperature, but more so to increases in stream discharge. For the present study, a temperature logger at each site would have been useful to determine if terrestrial drift activity was a function of daily or nightly air or water temperatures during the period each sampler was left out in the field.

Baxter Fausch et al. (2005) discuss the River Continuum Concept (Vannote et al., 1980) and comment that allochthonous inputs decrease with increasing stream size, the result of a decreasing ‘stream edge’ to ‘stream interior’ ratio (Naiman and Decamps, 1997). In the present study, larger stream sizes had reduced terrestrial drift densities (Figure 3-23 and Figure 3-24). This was particularly evident at Karekare and Opanuku streams that had the highest discharges (Figure 3-6) and widest stream widths (Figure 3-7), but also the lowest density of insects per m³ of water discharge (Figure 3-16). However, a larger stream surface area also allows more area for flying and falling insects to land in water and become entrained in drift (Lotrich, 1973), this may be especially so under forest riparian vegetation that provides habitat for insects and surfaces for them to fall from into stream drift. Figure 3-35 represents a theoretical relationship between terrestrial drift densities and stream width, where insect activity may initially increase or stay constant at small sized streams (dotted line). Stream width then reaches a threshold where the stream edge (riparian vegetation and stream perimeter) has less effect on the contribution of terrestrial insects to the stream as the stream gets larger in size.

![Theoretical diagram representing the amount of terrestrial insects in drift as streams get wider.](image)

The debris dams measured in this study differed in structural complexity, and this complexity dimension was not specifically quantified in the present study. Debris dams provide an increase in the faunal heterogeneity of a stream (Rolauffs et al., 2001). If a debris dam is structurally complex, it is possible that it may provide more water perimeter for terrestrial insects to enter stream drift.
from. The debris dam could then serve the dual purpose of not only providing habitat for stream fish (Rowe and Smith, 2003; Baker and Smith, 2007), but also increasing the chance that terrestrial insects on the debris may become entrained in drift.

3.7 SYNOPSIS OF KEY FINDINGS

- There were significantly higher densities of insects in drift during summer, compared to winter, during both day and night times.
- Significantly more aquatic drift occurred during the night compared to day.
- Overall, the study did not find unequivocal evidence to suggest that an increase in the amount of riparian cover will result in an increase in the amount of terrestrial insects in drift. However, with increasing riparian vegetation cover there was:
  - More insects in drift during night time in winter
  - Less terrestrial insects during winter
  - Less terrestrial insects by night during winter
  - Less aquatic insects during summer
  - Less aquatic insects by night during summer
  - Less aquatic insects by night during winter
- The number of terrestrial invertebrates present in stream drift increased with the amount of leaf detritus in drift.
- The study relied on correlation and regression values derived from a limited number of data points (8 for summer, 4 for winter) so the data should be interpreted with this in mind. The dependability of these conclusions would increase if a larger number of site replicates could be used, representing more data points in correlation and regression analysis.
- Close proximity vegetation cover had one significant linear relationship, this occurred with increasing amounts of terrestrial drift components during winter. However, employing the hemispherical photography technique to accurately quantify the proportion of close proximity cover would require further testing of the method.
- There is an information gap surrounding diurnal and seasonal variation in terrestrial drift from under evergreen forest vegetation typical of that found in New Zealand. This area could do with future research.
Further research into vegetation cover and contributions to the stream may like to consider a light estimation parameter to estimate vegetation density as a proxy for standing crop biomass of vegetation and potential above ground insect habitat. A method for estimation of cover proximity proportions may also be interesting future research.
4

PROVISION OF TERRESTRIAL DRIFT AND SUBSEQUENT CONSUMPTION BY BANDED KOKOPU.

4.1 INTRODUCTION

Invertebrates play a key role in stream food webs, providing a vital food source for some consumers (Cadwallader et al., 1980; Hansen et al., 2004; Allan and Castillo, 2007). These invertebrates originate from both in-stream and ex-stream sources. Understanding how the land-water interface functions will enable us to better comprehend how the contribution of energy from ex-stream sources determines ecosystem form and function within stream systems. This chapter describes the predator/prey interaction between banded kokopu and its terrestrially derived diet of invertebrates to determine the chapter objective:

*Does the diet of banded kokopu track prey item availability?*

Assessment of insect species data relationships provide the basis for comment on cause (insects provided) and effect (insects consumed) relationships between the prey opportunity and its subsequent consumption in the diet of banded kokopu. Assessing the dynamics of this interaction will enable us to better quantify how these fish depend on dietary resources obtained from both the terrestrial and aquatic environment and how these preferences vary both spatially and temporally. This chapter details results and interprets findings from insect capture and fish gut assessment to assess whether fish diet tracked prey availability.

4.2 RESULTS

The results presented describe aquatic macroinvertebrate composition as an autochthonous diet opportunity, drift insect composition as an allochthonous diet opportunity, and banded kokopu gut content species composition to assess fish preference for either type of prey. These results are statistically assessed and evaluated with multivariate techniques to classify and describe relationships that exist between sites and environmental parameters.
4.2.1 AQUATIC MACROINVERTEBRATES

There were 41.6% more aquatic macroinvertebrates in the benthos during summer than there were during winter. More aquatic macroinvertebrates occurred in the benthos at each of the summer sites compared to winter, however this was not a significant difference (Figure 4-1; \( n=4, p=0.08 \)). Most sites had a similar species composition, with the exception of Swanson Stream that had a large abundance of Diptera Simuliidae Austrosimulium (142 individuals) and Opal Pool during summer having large numbers of Amphipoda (78 individuals). Ephemoptera Larvae (\( \bar{x}=75.3 \)), Trichoptera Larvae (\( \bar{x}=43.3 \)) and Potamopyrgus snails (\( \bar{x}=38.8 \)) were common across all sites and all seasons. Figure 4-1b shows some inconsistent comparisons between summer and winter species abundances. Opal Pool exhibited a large number of Plecoptera during winter and a large number of Amphipods during summer.
Figure 4-1  Aquatic macroinvertebrate species: a) abundances and b) percentages from a composite kick sample at each site (n=1).
4.2.2 DRIFT INSECTS

Proportions of insects in terrestrial and aquatic drift (Figure 4-2) at most sites consisted largely of terrestrial insects, with the exception of Whakatai Stream during summer with a majority of aquatic components (82.5%). Three of the four sites showed a higher proportion of aquatic drift during winter (35-49%) compared to their summer stream counterparts (18-39%), while Whakatai Stream recorded large proportions of aquatic drift during summer (97.9%) and less during winter (0.43%). However, there was no significant difference in the proportion of aquatic components between summer and winter ($n=4$, $p=0.08$). The largest proportion of terrestrial insects compared to aquatic insects occurred at Swanson, Opanuku Tributary, Ocean View and Opal Pool during summer.

During winter, single samplers captured terrestrial drift in densities ranging from zero to 3.43 terrestrial insects/m$^3$ of water, while the corresponding summer sites had drift ranging from zero to 6.91 insects/m$^3$ of water. From all summer sites, drift ranged from zero to a maximum of 9.55 insects/m$^3$ of water.

![Figure 4-2 Percentage aquatic and terrestrial insects identified in drift per m$^3$ of water sampled across all sites ($n=18$).](image)

Ocean View exhibited the highest densities of insects in drift, with Opal Pool and Swanson also having large amounts of drift compared to the other sites (Figure 4-7a). Significantly more drift was captured during summer compared to winter sampling ($n=4$, $p=0.03$). During summer, the lowest drift densities occurred during summer at Opanuku (0.43 insects/m$^3$) and Karekare (0.50 insects/m$^3$) streams. These streams also had the widest stream channel (Figure 3-7) and the highest discharges (Figure 3-6) of all the summer sites. Opanuku Tributary had the least amount of insects...
captured in drift out of all the sites. There were significantly more terrestrial insects captured during summer than there were during winter ($n=24, p=0.00$) at the four sites.

Dipteran Flies were the largest contributors to insect drift across most sites (Figure 4-7b). Ants (Hymenoptera Formicidae), Ephemopteran Larvae, springtails (Arthropoda Collembola), Dipteran larvae and Potamopyrgus snails were also common in drift across all of the sites. There were 76% more insects available in drift during the summer than there were during winter.
Figure 4-3  Drift sample invertebrate species as a) count and b) proportion of insect species captured per m$^3$ of drift at each site (n=18).
The composition of samples collected during winter produced distinctly different communities of insects when compared to communities of insects captured during the summer, as shown by grouping number four in Figure 4-4. This indicates that temporal variation in community species composition and abundance is enough to distinguish distinct communities over any spatial distances that occur between sites. Opanuku and Karekare streams are distinctly different entities (grouping one and three), probably due to their uniquely different flow regimes and stream habitat causing species abundances to be relatively low compared to the other sites. Whakatai Stream also has a uniquely different species community in drift (grouping number five) due to its relatively larger proportion of Ephemoptera and Trichoptera Larvae (Figure 4-3). The uniqueness of Whakatai stream during summer is also shown by the large amount of aquatic insects identified from drift at that site during summer (Figure 4-2).

Increasing amounts of bank scarring, turbidity and increased temperatures were associated with winter sites, while increases in dissolved oxygen were associated with drift species composition at group two and group one.
Figure 4-4 Sites arranged by their composition of 89 species of drifting insect. Figure a): non-metric multidimensional scaled ordination plot with an environmental correlation overlay at an $r^2=0.2$ significance. Figure b): cluster dendogram, the blue line representing groupings at a defined 60% similarity.

### 4.2.3 BANDED KOKOPU

Fewer fish were caught at all of the sites that were sampled during winter compared to those same sites during summer (Figure 4-5), however this was determined non-significant difference ($n=4$, $p=0.2$). Small catches of fish occurred at Opanuku, Swanson, Waitoru Falls and Karekare Streams (1-5 fish). The higher number of fish caught at Opanuku Tributary, Ocean View, Opal Pool and Whakatai Streams (13-24 fish) provided a larger pool of fish data for a second sampling round to be based on. This, in part, justified the use of these sites for the winter sampling round.
There was a highly significant relationship between increasing conductivity and the number of fish caught ($n=12$, $r=0.74$, $p=0.01$), possibly indicating a relationship between low migration distances and increased densities of banded kokopu (Figure 1-5; Eikaas and McIntosh, 2006; Baker and Smith, 2007). In isolated cases, up to five banded kokopu were captured from single structurally complex debris dams. At a few sites during summer, the dermal surface of a few banded kokopu was infected with what looked like a dermal ‘white-spot’ fungal infection. On a few fish this infection was quite extensive and on one fish it covered approximately half of its body surface. During winter, this fungus was only evident on one fish. These infections may result from spawning related stressors and may be associated with seasonal mortality (Hardie et al, 2007). During summer a few fish were also observed to leak a white fluid from the anal area during manipulation which was possibly milt associated with spawning.

During electric fishing, longfin eels (*Anguilla diffenbachii*), koura (*Paranephrops planifrons*), crans bully (*Gobiomorphus basalis*) and shrimp (*Paratya curvirostris*) were commonly observed at all of the sites. Less commonly observed species were inanga (*Galaxias maculatus*), redfin bully (*Gobiomorphus huttoni*). Karekare stream also yielded two koaro (*Galaxias Brevipinnis*) and one shortjaw kokopu (*Galaxias postvectis*).

![Figure 4-5  Number of banded kokopu captured by electric-fishing. Error bars = ±1 standard error.](image)

Fish weight and length measurements adhered to a logarithmic curve as shown in Figure 4-6. No fish under 75 mm in length were measured in order to prevent damage during gut flushing. There were no significant differences between the weights ($n=64$, $p=0.51$) or lengths ($n=64$, $p=0.89$) of fish between summer and winter.
More insects were found in the guts of fish three sites during winter compared to summer. Waitoru Falls Stream fish had the highest number of insects in their gut per fish (12.6 insects, Figure 4-7), however only five fish were captured to obtain that result (Figure 4-5). There were significantly more invertebrate items in the guts of fish during winter ($\bar{x} =3.4$ insects/fish) compared to summer ($\bar{x} =5.6$ insects/fish; $n=64$, $p=0.00$). It is possible that the gut flushing technique utilised in this study became more refined after the first few sites in summer, resulting in higher numbers of insects obtained as the study progressed. However, this is unlikely as Waitoru Falls site was the first site studied and it returned the highest number of insects per fish gut.

Dissection of three individual fish after gut flushing, for gut content analysis showed the gut of one of the three fish to still contain a pill millipede (Arthropoda Diplopoda). This particular item was obviously small enough for the fish to consume, but was too large to be flushed back out of the fish's mouth after being gut flushed. Therefore, the gut flushing technique may under-represent the proportion of large insects with rigid body types that have difficulty being flushed back out of the fish gullet. It is also possible that items obtained from the fish gut that have a rigid body type (such as beetles or snails) are identifiable for a longer period of time after being consumed, more so than small items such as Dipteran flies and/or soft items such as mayflies. This may cause an under-representation of soft and/or small food items.

Fish dissection also indicated that items in the gut of the fish were mostly identifiable, but items in the post-gut intestinal tract were unidentifiable due to their already digested state. This confirms that the gut flushing technique was accessing the identifiable portion of all fish digestion tract...
components. Despite these limitations of the technique, the results obtained from gut flushing were satisfactory and the fish were able to be returned to the stream alive and unharmed. All fish recovered from the treatment/anaesthesia to a stable equilibrium swimming state in fresh stream water, the maximum recovery time taking only a few minutes.

![Figure 4-7 Average number of insects identified in the guts of banded kokopu. Error bars = ±1 standard error.](image)

4.2.4 FISH CONSUMPTION OF INSECTS

Aquatic and terrestrial components in the diet of banded kokopu varied between sites (Figure 4-8). Fish diet at all sites had a majority of terrestrial components consumed, when compared to the percentage of aquatic components. All sites sampled showed proportionately more terrestrial insects in diet during summer than those same sites during winter. Banded kokopu consumed significantly more terrestrial insects (\( \bar{x} = 79\% \)) than aquatic insects (\( \bar{x} = 21\% \)). There was no significant difference in the amount (\( n=4, p=0.69 \)) or proportion (\( n=4, p=0.06 \)) of terrestrial items in the fish guts between summer and winter.
Waitorus Falls Stream had the greatest number of items in the guts of fish during summer sampling (12.6 insects/fish), while Whakatai Stream had the least amount (2.2 insects/fish; Figure 4-9). Ocean View Stream had the highest amount of items in the fish gut from the four winter sites (8.7 insects/fish), while Opanuku Tributary had the least amount (3.3 insects/fish). There were significantly more contents in the guts of fish during winter at each site than there were in the guts of fish during summer at those four sites (n=64, p=0.00). Coleopteran Beetles and Hymenoptera Formicidae ants were common in the guts of fish across all sites and both seasons, while Trichoptera caddis larvae and Potamopyrgus snails were common in the guts during winter.

Ants (Hymenoptera Formicidae Sp.) and beetles (Coleoptera) comprise the largest percentages of banded kokopu diet. Wasps (Hymenoptera wasps), Water bugs (Hemiptera), cockroaches (Blattodea sp.), Caddis larvae (Trichoptera), aquatic snails (Potamopyrgus) and Mayfly larvae (Ephemoptera) also provided significant inputs across many sites. Fish diet between individuals was varied, with a few individual fish consuming large numbers of specific prey items. For example, four different fish consumed: 14 ants; 12 Potamopyrgus snails; 12 Trichoptera Conoecusidae Olinga; 10 Hemiptera Veliidae Microvelia.
Community analysis of site groupings determined from insect composition of fish gut contents are illustrated in Figure 4-10. Four distinct groupings occurred, when defined at 60% similarity. Opanuku showed a distinctly different gut species community than the other sites (grouping number 1). This occurred because gut content analysis at that site was only performed on one fish, which yielded three individual insects. Grouping number two yielded communities of insects in fish guts...
with high proportions of Hymenoptera Formicidae (Figure 4-9). Grouping number three contained three winter sites, all of which were close to the ocean and had relatively high proportions of aquatic insects in fish guts. Opal Pool (group four) was distinguished by its large portion of Hemiptera species. Of the environmental parameters, pH was strongly correlated with site distributions, while the stream vegetation cover parameter was not significantly correlated with any site distributions of the dataset.

![Figure 4-10](image)

Figure 4-10 Sites arranged by their composition of 47 insect species identified from banded kokopu gut contents. Figure a): non-metric multidimensional scaled ordination plot, with environmental correlation overlay at an $r^2=0.2$ significance. Figure b): cluster dendogram, the blue line representing groupings defined at a defined 60% similarity.
During summer, the relationships between sites according to the species identified in gut contents of banded kokopu are shown in Figure 4-11. Four distinct groupings have occurred when defined at 75% similarity. Group one contains Opanuku, which was defined by only one fish that was captured at that site, which explains why it has been classified as having the most distinct gut species composition. Group two contains five sites, all of which had fish with similar species of insects in their gut. Group three and four contain one site each, showing a unique composition of species in the guts of fish captured at those sites. Whakatai recorded the least vegetation cover and Opal Pool recorded the third least amount of vegetation cover from all the sites; however DIFN cover measurements were not a significant determinant of community composition.

The five sites in group two show gut species composition during summer to be so similar, that this multivariate analysis has placed each site directly on top of the others in that group. This indicates that during summer, banded kokopu at these sites are selective in their diet preferences, consuming a select group of insects, despite having the option to consume a broad range of invertebrates from their environment.
Assessment of the species community composition in the gut contents of all fish captured at the four winter sites, shows community composition to differ between the four sites, with a minor relationship between Whakatai and Ocean View streams (Figure 4-12a). However this relationship is not evident from the cluster dendogram of the same sites (Figure 4-12b). However, species community analysis of only four sites during winter gives only a limited basis of comparison to
assess multivariate relationships between these sites; therefore these results are likely to be subject to error due to the limited number of data points.

Figure 4-12  Sites arranged by the composition of 36 species identified in banded kokopu gut contents during winter. Figure a): non-metric multidimensional scaled ordination plot with an environmental correlation overlay at an $r^2=0.2$ significance. Figure b) shows a cluster dendogram.
4.2.5 FISH DIETARY PREFERENCES

Multivariate assessment of insect drift, after removal of species which banded kokopu did not consume are shown in Figure 4-13. Four distinct groupings were distinguished at 47% similarity; however these groupings were not the same as the site groupings presented in Figure 4-10 from the community analysis of gut contents. This suggests that the relationship between provision of food types in the environment of banded kokopu and the uptake of this food from drift is not solely defined by the density or type of insects provided. It is likely that a number of other factors interact to determine banded kokopu diet, such as; different fish and prey sizes, competition between fishes, habitat quality and seasonal activity regime of fishes.

The insect species available in drift for banded kokopu consumption at Whakatai Stream (Figure 4-13b, group four) arise from a distinctly different community of consumable insects compared to the other sites. This site is geographically isolated from any of the other sites, possibly explaining its unique composition of consumable insect types. Whakatai Stream also had higher conductivity, more debris dams and higher pH levels, which could also affect the composition of consumable insect species at that site. Figure 4-13 does not show any summer sites to pair closely with their respective winter site, again indicating that seasonal differences between sites better determine drift community composition over site specific factors.
Figure 4-13  Sites arranged by their composition of 47 species in drift that were a species consumed by banded kokopu. Figure a): non-metric multidimensional scaled ordination plot with an environmental correlation overlay at an $r^2=0.2$ significance. Figure b): cluster dendogram, the blue line representing community groupings at a defined 47% similarity.

Ivlev's electivity index in Figure 4-14 details the feeding preferences of banded kokopu. It must be noted that the electivity index does not handle well insects that have been provided in low numbers in drift or in the fish gut; therefore some numbers will be over-represented. For example, only one annelida (worm) was captured from all drift sampling and none of these worms were found in any fish gut contents, the electivity index however measured this as being a type of prey that was fully avoided by banded kokopu. Also, the nematode species obtained from fish guts were probably...
parasitic and actually living in the fish gut. Nevertheless, the rare species have still been included to comment on all species that were a component of banded kokopu diet.

The electivity index showed that the majority of species consumed by banded kokopu were terrestrial. Excluding rare species, the food items that were most strongly selected for were Hymenoptera wasps (0.89), Orthoptera wetas (0.85), Blattodea cockroaches (0.79), Coleoptera beetles (0.67), Arthropoda Psuedoscorpions (0.64) and Gastropod mollusca snails (0.60). In contrast, Paratya shrimp (-1), Mollusca limpet (-1) and Plecoptera larva (-1) were wholly selected against (excluding rare species). This provides evidence that the fish selectively consumes prey, as a number of species that were common in drift (Figure 4-3) were not found in the gut of fish; for example Hymenoptera flies, Dipteran flies and Arthropoda Collembola. These species are relatively smaller in size and were possibly not noticed by the fish, or were not worth the fish investing energy to seek out and consume them.

On the contrary, species strongly selected for were not often caught in drift, but were often found in gut contents as items favoured from drift by fish. Observational evidence suggested that commonly preferred items tended to be larger than those not commonly preferred. From all the species that were common and preferred (positive electivity in Figure 4-14), only Hemiptera and Psuedoscorpionida could be considered small relative to the others that were selected for. From the species that were not selected for (negative electivity in Figure 4-14), only Ephemoptera, Plecoptera and Mollusca Limpet could be considered large.
Figure 4-14 Ivlev’s Electivity Index of prey selection in the diet of banded kokopu from insects available in drift. An electivity index of -1 means the prey was present in the environment but was not consumed (wholly avoided). An electivity index measure of +1 means prey was present in the fish gut, but was not captured in the environment (wholly preferred). An electivity index of 0 means random feeding is occurring. * = species that had only a few individuals identified in both gut and/or drift samples.

When the electivity index (Figure 4-14) is separated into summer and winter components (Figure 4-15) the results differ slightly between seasons. From the aquatic species, none were preferred during summer; but during winter, Trichoptera caddis larvae (0.53), and Mollusca Potamopyrgus snails (0.69) were preferred (excluding rare species).
Figure 4-15  Ivlev’s Electivity Index of prey selection in the diet of banded kokopu from available insects and fish captured in a) summer (n=4) and b) winter (n=4). An electivity index of -1 means the prey was present in the environment but was not consumed (wholly avoided), conversely, an electivity index measure of +1 means the prey was present in the fish gut but not captured in the environment (wholly preferred). An electivity index of 0 indicates a species captured in drift and fish gut contents in equally proportionate amount (randomly consumed). * = rare species that had only a few individuals identified in both gut and/or drift samples.
Despite a classification into small and large size classes (Table 4-1), terrestrial drift components consistently dominated the gut contents of banded kokopu at both 75-125mm (85.82%) and at 125mm+ (83.50%) size classes. This shows that similar percentages of terrestrial components occurred in fish diets despite their differing size classes (above 75 mm in length).

Table 4-1  Percentage terrestrial and aquatic dietary components of banded kokopu of less than 125mm length (small), over 125mm length (large), and for all fish combined. Fish under 75mm in size were not sampled.

<table>
<thead>
<tr>
<th></th>
<th>75-125mm</th>
<th>&lt;125mm</th>
<th>All Fish</th>
</tr>
</thead>
<tbody>
<tr>
<td>Terrestrial</td>
<td>85.82</td>
<td>83.50</td>
<td>84.80</td>
</tr>
<tr>
<td>Aquatic</td>
<td>14.18</td>
<td>16.50</td>
<td>15.20</td>
</tr>
</tbody>
</table>

Linear regression equations were run to directly assess abundances of insects in drift against those insects that were actually consumed by banded kokopu. Only one statistically significant linear relationship could be determined, as shown in Table 4-2, between aquatic invertebrates present in drift and the aquatic invertebrates consumed by banded kokopu during summer. This relationship shows that aquatic invertebrate species abundances explained 12% of the variability in the aquatic component of banded kokopu diet. The lack of other regressive relationships indicates that consumption of aquatic and/or terrestrial invertebrate prey was subject to other key causal mechanisms that caused variation in banded kokopu diet.

Table 4-2  Regression significance values for relationships between species of invertebrate in banded kokopu diet and invertebrate drift classified by the headings below. Data in bold show a statistically significant relationship at the 99% threshold.

<table>
<thead>
<tr>
<th></th>
<th>Aquatic and Terrestrial</th>
<th>Terrestrial</th>
<th>Aquatic</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Summer (8 sites) Winter (4 sites)</td>
<td>All (12 sites)</td>
<td>Summer (8 sites) Winter (4 sites)</td>
</tr>
<tr>
<td>n</td>
<td>191 94 287 127 63 191</td>
<td>63 31 95</td>
<td>0.00 0.40 0.14</td>
</tr>
<tr>
<td>p</td>
<td>0.16 0.21 0.09 0.33 0.44 0.20</td>
<td>0.35 0.15 0.15</td>
<td></td>
</tr>
<tr>
<td>r</td>
<td>0.10 0.13 0.10 0.09 0.10 0.09</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
### 4.2.6 VEGETATION COVER AND BANDED KOKOPU DIET

Simple linear regression to assess the relationship between stream DIFN cover and the proportion of terrestrial or aquatic components in the gut of banded kokopu did not uncover any significant relationships between the two variables Table 4-3.

Table 4-3  Regression significance values for the relationships between terrestrial or aquatic items in banded kokopu diet and stream DIFN cover.

<table>
<thead>
<tr>
<th></th>
<th>Terrestrial and Stream DIFN</th>
<th>Terrestrial and Bank DIFN</th>
<th>Aquatic and Stream DIFN</th>
<th>Aquatic and Bank DIFN</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Summer (8 sites)</td>
<td>Winter (4 sites)</td>
<td>Summer (8 sites)</td>
<td>Winter (4 sites)</td>
</tr>
<tr>
<td><strong>n</strong></td>
<td>8</td>
<td>4</td>
<td>8</td>
<td>4</td>
</tr>
<tr>
<td><strong>p</strong></td>
<td>0.17</td>
<td>0.75</td>
<td>0.15</td>
<td>0.55</td>
</tr>
<tr>
<td><strong>r</strong></td>
<td>0.55</td>
<td>0.25</td>
<td>0.56</td>
<td>0.45</td>
</tr>
</tbody>
</table>

Diet composition data presented in Figure 4-11 during summer and Figure 4-12 during winter provide little evidence to suggest a relationship between community groupings of the gut insects identified and vegetation cover over the stream.

Assessment of regression relationships between fish diet, densiometer and hemispherical photography cover measurements provided only one significant relationship (Appendix 0, Table 6-3). This relationship was present between the densiometer cover measurement and the amount of aquatic insects in the diet of banded kokopu during summer \((n=8, \ p=0.01, \ r=0.83)\). However, given that densiometer cover measurements, stream widths and discharges during summer were mutually dependant (Table 3-12), it is likely that these width and discharge factors provided more influence on aquatic macroinvertebrate populations than did stream cover.

### 4.3 DISCUSSION

This section is compiled to best consider the factors that collectively combine to influence banded kokopu diet composition, providing discussion to evaluate if fish diet tracks terrestrial prey availability.
4.3.1 METABOLIC ACTIVITY REGIME AND SEASONAL EFFECTS ON DIETARY UPTAKE

In the present study, considerably less aquatic and terrestrial insects were available for banded kokopu consumption during the winter when compared to summer. The banded kokopu life cycle may have adapted to this in response to an annual deficiency in prey opportunities during winter by reducing its metabolic activity, slowing gastric evacuation rates and lowering energy demand during this time (West et al., 2005). This situation would explain how banded kokopu had more insects in their gut during winter at three of the four sites (excluding Ocean view; Figure 4-9), despite there being less benthic aquatic (Figure 4-1) and less drifting invertebrates (Figure 4-3) available during winter. The fish may feed more passively to conserve energy, consuming insects in the water column as they drift toward the fish rather than actively seeking them out. Energy may then be reserved for taking shelter and retaining position in the stream during formative flood events (Rowe and Smith, 2003) in winter.

If banded kokopu adhere to the strict definition of ‘nocturnal’ activity and feeding periods (McDowall, 1990) then they have extended periods of feeding time during the longer winter nights. Although both aquatic and terrestrial prey densities are lower, longer feeding periods give the fish more opportunity to gather the food it requires for sustenance. However, the effects of extended foraging hours are largely negated if the majority of their diet is consumed during crepuscular peaks in foraging activity (McCullough, 1998; McCullough and Hicks, 2002), as these periods will occur in roughly equal proportions during both summer and winter.

During summer, when terrestrial drift rates increase (see section 3.6.3) banded kokopu may selectively consume terrestrial invertebrates in drift, while their consumption of aquatic macro-invertebrates may become more opportunistic, being consumed only if they drift into the detection path of the fish.

4.3.2 OPPORTUNITY AND UPTAKE

Banded kokopu prefer slow flowing pools within small streams (McCullough, 1998) that best facilitate lateral line feeding (Halstead, 1994; Baker, 2000). This style of feeding ensures they expend less energy and can feed from a greater concentration of insects that occur in drift compared to a stream of larger size (see section 3.6.5). Considering this, although the densities of insects in drift were lower during winter, stream discharge was higher, therefore there were more insects flowing down the stream per minute than the insect density measurement alone can communicate (Table 3-16). A larger proportion of net insects available in drift with increasing stream discharges help to explain how banded kokopu cope with such a reduction in insect densities during winter.
Although it will be more difficult to catch the insects due to the lower densities per cubic meter of water, there are more cubic meters of water for banded kokopu to search through during winter when discharges are higher.

Annual aquatic invertebrate abundances are variable, depending on environmental stresses the stream has undergone preceding sampling, however macroinvertebrate abundances overall tend to fluctuate around a steady state (Harding et al., 2004). Terrestrial insect subsidies are highly variable over space and time (Nakano et al., 1999b; West et al., 2005) due to the heterogeneous nature of terrestrial environments. This was evident when assessing high numbers of wasp counts from stream drift during summer, suggesting a subsidy pulse contribution to the stream when wasps populations die at the end of summer and autumn (Barlow et al., 2002). Wasps were however almost non-existent in winter drift samples in the present study. This variation in terrestrial input was also evident in drift samples, which ranged from a minimum of zero terrestrial insects, to a maximum recorded terrestrial insect density of 9.55 insects/m$^3$ of water for any day or night sampling period.

Given that allochthonous resources provide energy to the stream over and above constantly available within stream energy sources, terrestrial components may define animal population dynamics better than autochthonous stream resources (Kawaguchi and Nakano, 2001). However, increased banded kokopu densities with increased densities of autochthonous organisms (West et al., 2005) provide evidence to suggest that the inverse occurs. Aquatic macroinvertebrates may better define trophic population dynamics as a reliable basal resource for banded kokopu populations, despite their preference for terrestrial items.

This relationship has been further defined by Nakano, Myasaka et al. (1999b) who conclude that in streams containing insectivorous fish, allochthonous resources best determine animal population dynamics (trophic cascades) during summer. However, despite autochthonous prey being a minority component of banded kokopu diet, the present study provided evidence to suggest that aquatic prey consumption does track aquatic prey availability during summer. Aquatic macroinvertebrates as a basal resource could provide security for banded kokopu during times of dietary hardship such as that observed during winter when diet opportunities are low and during summer when energy demands are larger and consumption demand increases along with total invertebrate densities. This idea is one consistent with optimal foraging theory (Ostfeld, 1982; Barnes and Mann, 1991; Dodds, 2002). An exclusion experiment similar to that of Nakano et al. (1999b) could be set up to test if the banded kokopu exhibit this feeding strategy. The experiment by Nakano et al. demonstrated that salmonid fishes switched their prey preferences towards aquatic arthropods in the absence of terrestrial arthropods.
The proportion of terrestrial drift in banded kokopu diets increase when the density of allochthonous insects in drift increase (West et al., 2005). This is evidence to suggest that prey uptake does track prey opportunity. However, this relationship was not clearly defined as being significant in the present study, especially where regression relationships between supply and uptake were not significant (Table 4-2).

The present study allowed fish to feed from drift while drift sampling was taking place, allowing for calculation of dietary preference by way of Ivlev's electivity index. However, exclusion of fish from foraging in the drift sampling zone would allow for regression type equations to be calculated that directly regress provision (dependant) to uptake (independent). Not excluding fish from feeding during drift sampling causes an under-representation of preferred items, causing error in regression calculations.

 Preferential prey selection must also be considered when assessing insect consumption responses to changes in supply. Diet composition in the present study varied between individuals at a site, but averaged across the site, defined a select group of preferred dietary items (e.g., Figure 4-11). Determination of prey consumption from prey opportunity would be more appropriately framed with some consideration to a fish's preference for larger prey items in terrestrial drift.

4.3.3 PREY SIZE PREFERENCES - NUMERICAL AND GRAVIMETRIC ASSESSMENT

The present study only quantified fish diet preferences numerically, but gravimetric analysis is clearly important in the assessment of fish diet preference. Main and Lyon (1988) ascertained that banded kokopu preferentially select larger individuals of beetle to consume, a finding similar to that of West, Jowett et al. (2005) who also found larger invertebrate items to be preferentially selected. Other fish such as drift feeding trout (Allan, 1978; Nakano et al., 1999a) and shortjaw kokopu (Galaxias postvectis) have also been found exhibit a preference for larger sized prey items.

Main and Lyon (1988) further determined that beetles contributed 57.5% of gut content biomass. Consumption of larger insects is advantageous, due to the smaller proportion of insect exoskeleton as a function of increasing insect body mass (Edwards and Huryn, 1996), providing a significant energy contribution to the fish for the relatively little energy expenditure taken to capture and consume the individual. The diet preferences of banded kokopu seem to be simply defined by economy, obtaining the largest energy gain for the least amount of effort. An equation simplified by Allan (1995) as:

\[
\text{Energy Gain} = \frac{\text{Dry mass or calories obtained}}{\text{Energy expended in prey acquisition}}
\]
This should also hold true for larger insects consumed by banded kokopu in the present study, such as beetles, wasps, cockroaches, wetas, centipedes and millipedes. Main and Lyon (1988) and West et al. (2005) detail that gravimetric proportions (90.9%; 89%) greatly exceed that of numerical proportions (68.9%; 75%) of terrestrially derived invertebrate taxa in banded kokopu gut contents. West, Jowett et al. (2005) further show the average weight of a terrestrial item in their study was 4.5mg, compared to 1.5mg for an aquatic item in the diet of banded kokopu.

In the present study, ants were smaller in size than beetles and wasps consumed by banded kokopu, despite this, ants exhibited a similar electivity preference. This is probably explained by the ants’ disruption of water surface tension, triggering the capable motion detection and olfactory senses of banded kokopu (Halstead, 1994; Baker, 2000). Banded kokopu will likely prefer larger prey, unless smaller size prey attracts their attention by disrupting water surface tension, increasing the insects’ chances of detection.

It is possible that consumption of small insects (such as Dipteran flies often present in relatively high numbers compared to other terrestrial insects) is not worth the energy investment for banded kokopu when other, more profitable energy sources are available. However, dietary stress may cause a behavioural adaptation to actively seek out forms of prey that are not so readily accessible, such as smaller insects or aquatic invertebrates that are more difficult to capture.

Numerical results of the present study are only indicative of the time that samples were taken and do not provide information on annual, inter-seasonal or other local scale fluctuations in fish diet preference. In a study by West et al. (2005), terrestrial prey was the lowest in spring and highest in autumn, providing evidence to suggest that summer and winter did not provide the highest and lowest densities of drift insects annually (Table 4-4). Further study could assess how changes occur in invertebrate drift and fish diet occur over longer or more regular time periods, gaining a more accurate indication of insect consumption adjustments throughout the year.

4.3.4 INTER AND INTRA-SPECIES SPECIFIC COMPETITION

Fish diet can also be affected by competition within species and competition between species. Competition with conspecifics can limit the amount of terrestrial drift available for other individual banded kokopu. Evidence of this density dependant interaction has been shown in West et al. (2005), where increased banded kokopu abundances slowed fish growth rates, restricted movement between pools and increased aggressive interactions between banded kokopu, causing higher amounts of fin damage as fish densities increase.
Seemingly contradicting West’s findings that high fish densities can retard growth effects are the findings of Rowe and Smith (2003), who ascertain that as banded kokopu relative abundance increases, so do fish sizes. Rowe and Smith also detail that increased habitat quality is then related to high banded kokopu abundances. After assessing the findings of these two studies, one could postulate that in Rowe and Smith’s study, banded kokopu abundances were limited by the amount of available habitat. West’s’ banded kokopu populations had abundant ideal habitat and the highest recorded fish densities to date (in 2005), and were limited by the amount of diet available, causing competitive interactions for food.

If banded kokopu exist in pools immediately upstream of a fishes’ drift collection area, the fish below either rely on fish upstream not consuming all of the insects passing through their pool, or they rely on a contribution of insects to the stream between pools, or a mixture of the two (Hansen et al., 2004). This assumption of diet limitation due to competition for dietary resources would be a function of the spatial distribution and density of fishes along the stream reach. If dietary stress is caused due to a lack of diet opportunities because other fish are consuming most of the available food, then fish may adopt a behavioural adaptation to cater for the lack of dietary intake (Nakano et al., 1999b).

An adaptation mechanism such as this can be the result of dominance hierarchies existing between and within pools where high numbers of banded kokopu are present (e.g., David and Stoffels, 2003). High densities of fish may cause individuals to consume varied size and/or species fraction of invertebrates from that wholly available. Dominant fish then obtain exclusive feeding rights from all insects that enter that pool, or a prime feeding position within a pool that many fish occupy. However, little research has been undertaken on these postulated banded kokopu interaction behaviours with conspecifics. Dominance hierarchies have been shown to exist among primarily nocturnal giant kokopu (Galaxias argentus) populations where larger fish force smaller subordinate fish to feed by day (David et al., 2007), temporally and spatially altering their feeding behaviour (Hansen and Closs, 2005). However, in the present study, separation of banded kokopu into size fractions did not alter the proportions of terrestrial components in diets (Table 4-2). It is likely that competition with other fish is not a significant factor that determines banded kokopu diet in streams in the present study, when comparing these fish densities to that observed in a very small ephemeral urban stream in Auckland, where numbers of various sized banded kokopu existing in pools far exceed those densities observed in the present study. It may be that a size related dominance hierarchy is density dependant and is only established when a) a certain fish density is reached or b) when residential habitats, such as a dimensionally complex debris dam, necessitate these competitive interactions between fish that reside amongst this type of key habitat.
Another factor that could cause a widely varied diet of banded kokopu along a stream reach is variable micro-scale donations of terrestrial insects in response to variable habitat features. Vegetation, bank morphology, debris dams and stream orientation are examples of micro-habitat features that constantly change throughout a study reach, all altering donor habitats from which terrestrial insects could be donated to the stream.

Banded kokopu diet could also be variable due to diet exclusion effect caused from other species that live in the same stream and compete for similar food resources (e.g., McIntosh et al., 1992). Longfin eels (*Anguilla diffenbachii*) existed at all study sites in low densities and could be a dietary competitor, considering that they consume a range of insects similar to that of banded kokopu (Hicks, 1997). However, longfin eels are more generalist feeders (McDowall, 1990; McDowall, 2000) and existed in lower densities than banded kokopu at study sites, perhaps being more transient along the stream in search of their food than banded kokopu.

### 4.3.5 COMPARISON TO PREVIOUS STUDIES OF BANDED KOKOPU DIET

A study undertaken by Main and Lyon (1988) under native forest in South-Westland showed that proportions of terrestrial taxa had less variation between seasons than the present study (Table 4-4). Our results show higher abundances of terrestrial activity in drift to occur during the summer (87.6%) and a lower proportion of terrestrial taxa in drift during the winter (58.7%) in the Waitakere Ranges. Main and Lyon (1988) found the opposite to occur, with higher abundances of terrestrial taxa consumed during winter (71.3%) compared to summer (68.1%) in South Westland, however they do not explain whether abundances of terrestrial drift items available were higher during summer or winter. The present study found that the abundance of all invertebrate drift was reduced during winter and that higher abundances of terrestrial taxa occurred in drift during summer. The three studies presented in Table 4-4 show that the proportion of terrestrialially derived invertebrates in banded kokopu diet from all three studies was comparatively similar. This finding is consistent with multivariate analysis presented in Figure 4-11, where gut content analysis showed species assemblages to be almost identical for five of the eight study reaches during summer. This result suggests that despite spatial isolation, banded kokopu consumed a similar range of insects during summer.

Main and Lyon (1988) found that the terrestrial diet of banded kokopu consisted largely of Coleoptera, followed by Hymenoptera, then Arachnida. The results obtained in this present study also show Coleopteran beetles and Hymenopteran ants to be major dietary contributors, followed by Hemipteran water bugs, Potamapyrgous snails, Hymenopteran wasps, and Trichopteran caddis larvae.
Table 4-4  Comparison of the numerical proportion of terrestrial components (as opposed to aquatic components) found in the gut of banded kokopu.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Summer</td>
<td>68.1 %</td>
<td>76.6 %</td>
<td>87.6 %</td>
</tr>
<tr>
<td>Winter</td>
<td>71.3 %</td>
<td>74.8 %</td>
<td>58.7 %</td>
</tr>
<tr>
<td>Average</td>
<td>69.7%</td>
<td>75 % (four seasons)</td>
<td>73.1%</td>
</tr>
</tbody>
</table>

4.4 SYNOPSIS OF KEY FINDINGS

- To answer the chapter objective, there was little evidence to suggest that banded kokopu diet directly tracked terrestrial prey abundances. However:
  - Banded kokopu were selective in their eating habits and appeared to prefer insects that were gravimetrically larger.
  - Banded kokopu in the Waitakere Ranges consumed a consistently similar community of species across most sites during summer.
  - Banded kokopu consumed more aquatic insects when more were available during summer.
  - Insect drift should be assessed in terms of density and net movement, due to variation in insect opportunity during fluctuating stream discharges.
  - Banded kokopu had more prey items in their guts during winter than they did during summer.
- More research could be undertaken to better understand social hierarchies and competitive behaviour interaction within banded kokopu populations, allowing a better understanding of what causes variation in fish diets.
- Further study could gain a more accurate assessment of how terrestrial drift and/or fish diet varies throughout the course of the year at a finer resolution than just summer and winter comparisons presented in this study.
Chapter 5 | P.G. Hancock

5 CONCLUSIONS

5.1 VEGETATION COVER, INSECT AVAILABILITY AND FISH DIETARY UPTAKE

The work in this thesis examined two key components. The first aimed to determine if an increase in the amount of riparian cover in evergreen forest systems causes a subsequent increase in the amount of insects becoming trapped in stream drift (Objective 1 in Figure 5-1). This should then result in those insects becoming available for insectivorous fish to consume. Terrestrial drift abundances were variable over time and between study sites. The relationship between terrestrial insect densities and environmental drivers is complex and determining the extent that one of these drivers (such as vegetation cover) affects insect communities is difficult. In the present study, no direct causal relationship was observed between the extent of vegetation cover and allochthonous insect abundance in drift. A number of relationships were however observed when insect abundance data was analysed at a finer seasonal and/or diurnal resolution (Section 3.7).

The second key component in this thesis investigated how banded kokopu diet responded to variation in the amount and type of terrestrial insects presented to them amongst stream drift (Objective 2 in Figure 5-1). Banded kokopu tended to be quite selective in their prey consumption from the range of insects that were wholly available to them. Given that banded kokopu exist in other streams at much higher densities (e.g., West et al., 2005) than the streams utilised in this present study and other studies (e.g., Rowe and Smith, 2003), it is likely that availability of food was not a limiting factor for banded kokopu in this present study. Banded kokopu existing in the Waitakere study streams could most likely afford to selectively consume from the items that were available to them in drift. However, a lack of feeding pools of optimum size and flow to facilitate lateral line feeding, and lack of residential habitat are likely to be the major habitat quality factors that limit banded kokopu abundances in the Waitakere Ranges. Although this study did not intentionally investigate fish abundances or densities, consideration of this factor could help better understand fish eating habits.
To investigate the direct interaction between vegetation cover and fish diet, a second hypothesis (Hypothesis 2 in Figure 5-1) determined if a direct causal relationship existed between the amount of canopy cover and fish diet. With reference to insectivorous fish, England et al. (2004) found that small reductions in forest cover cause reductions in the stream food web’s dependence on terrestrial subsidies. In studies of amphibious salamanders, Greene et al. (2008) found that late successional forests had less terrestrial prey, causing salamanders to move further from the stream in search of food. Despite this evidence of cover variation affecting insect abundances, the direct relationship between these two factors was not observed in this present study. This was possibly due to the amount of other environmental pressures (e.g., coastal and inland vegetation types, air temperatures, stream flow velocities) that combined to influence both insect drift densities and banded kokopu prey selection.
5.2 KNOWLEDGE LIMITATIONS AND SUGGESTED RESEARCH

Future work is warranted on habitat and density dependant limitations of banded kokopu populations. This would better determine the point at which habitat no longer restricts banded kokopu numbers and if a subsequent increase in competitive interactions occur when densities increase towards the carrying capacity (k). Fish diet may differ between habitat limited streams with low fish densities, and optimal habitat streams that are only limited by diet availability at high fish densities.

A fish exclusion experiment similar to Nakano et al. (1999b) could be set up to test whether lack of terrestrial prey for banded kokopu would cause a shift in prey preferences, consistent with optimal foraging theory. This type of research could reveal how banded kokopu adapt to a limitation in prey availability at each stream.

A future study on the gastric evacuation rates of banded kokopu during different times of the year could help indicate how long gut contents are held, to later infer rates of metabolic activity. This information would help explain feeding rates of banded kokopu from the average number of items obtained from gut flushing at different times of the year.

Future work could be undertaken to define an analytical technique that determines vegetation cover percentage proximity to the stream. Different proximity components of vegetation cover in a forest ecosystem may provide differential contributions to the stream system. It may be that close proximity vegetation provides a higher proportion of terrestrial insects to stream drift per unit leaf area, than does far proximity vegetation. A future study could possibly compare the ecological properties of four cover types: Forest vegetation (100% cover, no understory (possibly grazed)), scrub/grass vegetation cover (no forest overstory, 100% cover), both overstory and understory vegetation (100% cover) and an open site (as a control, no/little vegetation cover). This study could be designed to test cover proximity and might compare the functional difference between riparian vegetation types, with the a-priori hypotheses that:

- a) Large overstory vegetation is a better stream cooling mechanism than small scrub/grass vegetation cover, with a combined overstory and understory vegetation treatment providing optimum stream cooling conditions.

- b) Small vegetation cover is a better contributor of terrestrial invertebrates to the stream than large vegetation cover, with a combined understory and overstory treatment providing the optimum terrestrial invertebrate contribution conditions.
5.3 SYNOPSIS

- This study could not find conclusive evidence to suggest that increasing vegetation cover (>65% cover) caused an increase in the abundance of terrestrial invertebrates drift.
- Banded kokopu in this study were selective in their feeding habits and did not feed at random from the total available portion of drifting invertebrates.
- Comparison to other studies suggests that abundant food was available to fish in streams utilised in this study. However, a lack of optimal feeding habitat and residential habitat restricted the abundance of banded kokopu at these locations.
- Densities of insects in drift were greatly increased during summer compared to winter.
- Drift assessment in relation to fish diet should consider assessment of drift density and also net drift movement due to flow rate fluctuations.
- Further research could assess patterns of diurnal and seasonal variation in terrestrial drift from under evergreen forest vegetation typical of that found in New Zealand.
### 6.1 SPECIES LIST

Table 6-1 List of all invertebrate species identified in this study, their common name and their categorisation in either aquatic or terrestrial groupings. Note: Some unidentified species are included at the end.

<table>
<thead>
<tr>
<th>Taxonomic Name</th>
<th>Common name</th>
<th>T=Terrestrial</th>
<th>A=Aquatic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Diptera Larval/Pupa Sp.</td>
<td>Flies</td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Diptera Fly Sp.</td>
<td>Flies</td>
<td></td>
<td>T</td>
</tr>
<tr>
<td>Diptera Simuliidae Austrosimulium</td>
<td>Sand fly</td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Diptera Dixidae Paradixia</td>
<td></td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Diptera Dixidae Nothodixia</td>
<td></td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Diptera Tanyderidae Mischoderus</td>
<td></td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Diptera Chironominae</td>
<td></td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Diptera Nematocera</td>
<td></td>
<td></td>
<td>A</td>
</tr>
<tr>
<td>Ephemoptera Larval/Pupa Sp.</td>
<td>Mayflies</td>
<td></td>
<td>A</td>
</tr>
<tr>
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<td>Mayflies</td>
<td></td>
<td>T</td>
</tr>
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<td></td>
<td>A</td>
</tr>
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<td></td>
<td>A</td>
</tr>
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</tr>
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</tr>
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<td>A</td>
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</tr>
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<td></td>
<td>A</td>
</tr>
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<td>Trichoptera Hydropsychidae Aoteapsychae</td>
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<td></td>
<td>A</td>
</tr>
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<td>Trichoptera Hydropsychidae Oxyethira</td>
<td>Gills caddis</td>
<td></td>
<td>A</td>
</tr>
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<td></td>
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<td>Caddis in fine sand case</td>
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<tr>
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<td>Secreted case caddis</td>
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<td></td>
</tr>
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<td>Trichoptera Philopotamidae Hydrobiosella</td>
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<td>A</td>
<td></td>
</tr>
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<td>Trichoptera Philopeithridae Philorheithrus</td>
<td></td>
<td>A</td>
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<td></td>
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<td>A</td>
<td></td>
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<td>Trichoptera Polycentropodidae Plectrocnemia</td>
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<td>A</td>
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<td>Stoneflies</td>
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<td>A</td>
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</tr>
<tr>
<td>Plecoptera Austroperlidae Austroperla cyrene</td>
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<td></td>
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<tr>
<td>Plecoptera Antarctoperlinae Zelandiobius</td>
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<td>A</td>
<td></td>
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<tr>
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<td></td>
</tr>
<tr>
<td>Plecoptera Gripopterygidae Megaleptoperla</td>
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<td>Beetle</td>
<td>T</td>
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</tr>
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<td>Coleoptera Larvae sp.</td>
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<td>Coleoptera Hydrophilidae</td>
<td></td>
<td>A</td>
<td></td>
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<td>Coleoptera Ptilodactylidae</td>
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<td>Coleoptera Hydraenidae</td>
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<td>Coleoptera Elimidae</td>
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</tr>
<tr>
<td>Hymenoptera sp.</td>
<td>Hymenoptera flies</td>
<td>T</td>
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<td>T</td>
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<td>German wasp</td>
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<td>Hemiptera Sp.</td>
<td>Water bugs</td>
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<td></td>
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<td>Athoracophoridae</td>
<td>Leaf veined slug</td>
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<td></td>
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<td>Blattodea Sp.</td>
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<td>Hemiptera Veliidae Microvelia macgregori</td>
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<td>Hemiptera Mesoveliidae</td>
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</tr>
<tr>
<td>Hemiptera Salidae</td>
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<tr>
<td>Lepidoptera sp.</td>
<td>Moth</td>
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</tr>
<tr>
<td>Orthoptera sp.</td>
<td>Weta</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Chilipoda</td>
<td>Centipede</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Diplopoda</td>
<td>Millipede</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Arachnida Araneida</td>
<td>Spiders</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Arachnida Acaridae</td>
<td>Mites</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Arachnida Pseudoscorpionida</td>
<td>Pseudoscorpions</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Collembola</td>
<td>Springtail</td>
<td>A</td>
<td></td>
</tr>
<tr>
<td>Arthropoda Crustacea Isopoda Oniscoidea</td>
<td>Slaters</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Mollusca Limpet</td>
<td>Limpet</td>
<td>A</td>
<td></td>
</tr>
<tr>
<td>Mollusca Gastropoda</td>
<td>Small Snail</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Homoptera</td>
<td>Cicadia, leafhopper, scale</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Amphipoda</td>
<td>Hoppers</td>
<td>T</td>
<td></td>
</tr>
<tr>
<td>Gastropoda Mollusca Potamopyrgus</td>
<td>Freshwater snail</td>
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<td>Megaloptera Corydalidae Archichauliodes diversus</td>
<td>Dobsonflies</td>
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<td></td>
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<tr>
<td>Parastacidae Paranephrops planifrons</td>
<td>Koura</td>
<td>A</td>
<td></td>
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</tbody>
</table>
### 6.2 STREAM DIFN COVER AND DRIFT INSECT REGRESSION GRAPHS

Table 6-2 Regression relationships between stream DIFN cover and insects in drift.

<table>
<thead>
<tr>
<th>Sample specimen 1</th>
<th>Sample specimen 2</th>
<th>Sample specimen 3</th>
<th>Sample specimen 4</th>
<th>Sample specimen 5</th>
<th>Sample specimen 6</th>
<th>Sample specimen 7</th>
<th>Sample specimen 8</th>
<th>Sample specimen 9</th>
<th>Sample specimen 10</th>
<th>Sample specimen 11</th>
<th>Sample specimen 12</th>
<th>Sample specimen 13</th>
<th>Sample specimen 14</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dipteran Fly T</td>
<td>? T</td>
<td>Millipede A</td>
<td>Dipteran Larvae A</td>
<td>Beetle T</td>
<td>Dipteran Fly T</td>
<td>Dipteran Fly T</td>
<td>Trichoptera Fly T</td>
<td>Beetle T</td>
<td>Beetle A</td>
<td>Springtail A</td>
<td>Dipteran Larvae A</td>
<td>Hopper T</td>
<td>Hemipteran T</td>
</tr>
</tbody>
</table>

#### Summer

![Stream DIFN Cover and Insects in Drift during Summer](image)

**DF=6, r=-0.14, p=0.74**

#### Winter

![Stream DIFN Cover and Insects in Drift during Winter](image)

**DF=2, R=-0.87, p=0.12**

---

<table>
<thead>
<tr>
<th>Atyidae Paratya</th>
<th>Shrimp</th>
<th>A</th>
</tr>
</thead>
<tbody>
<tr>
<td>Annelida</td>
<td>Worm</td>
<td>T</td>
</tr>
<tr>
<td>Isoptera Kalotermitidae</td>
<td>Termite</td>
<td>T</td>
</tr>
<tr>
<td>Nematoda</td>
<td>A</td>
<td></td>
</tr>
</tbody>
</table>

---

Atyidae Paratya | Shrimp | A |
Annelida        | Worm   | T |
Isoptera Kalotermitidae | Termite | T |
Nematoda        | A      |   |
Terrestrial

DF = 6, r=0.07, p=0.87

Aquatic

DF=6, r=-0.61, p=0.11

Day

DF=6, r=-0.17, p=0.68
Night

DF=6, r=-0.08, p=0.85

DF=2, r=-0.91, p=0.08

Terrestrial - Day

DF=6, r=-0.91, p=0.83

DF=6, r=-0.85, p=0.15

Terrestrial - Night

DF=6, r=0.22, p=0.60

DF=2, r=-0.95, p=0.05
Aquatic - Day

DF=6, r=-0.40, p=0.33

Aquatic - Night

DF=2, r=-0.67, p=0.07

DF=2, r=-0.98, p=0.11
### 6.3 Regression Relationships between Banded Kokopu Diet and Densiometer and Hemispherical Photography Vegetation

Table 6-3 Regression relationships between components in the diet of banded kokopu and densiometer and hemispherical photography cover measurements. Numbers in bold represent a statistically significant relationship at the 99% threshold.

<table>
<thead>
<tr>
<th></th>
<th>Terrestrial and Densiometer</th>
<th>Terrestrial and Hemispherical Near Cover</th>
<th>Aquatic and Densiometer DIFN</th>
<th>Aquatic and Hemispherical Near Cover</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Summer (8 sites)</td>
<td>Winter (4 sites)</td>
<td>Summer (8 sites)</td>
<td>Winter (4 sites)</td>
</tr>
<tr>
<td><strong>n</strong></td>
<td>8</td>
<td>4</td>
<td>8</td>
<td>4</td>
</tr>
<tr>
<td><strong>p</strong></td>
<td>0.38</td>
<td>0.97</td>
<td>0.40</td>
<td>0.26</td>
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<tr>
<td><strong>r</strong></td>
<td>0.36</td>
<td>0.00</td>
<td>0.12</td>
<td>0.74</td>
</tr>
</tbody>
</table>
AQUI-S New Zealand Ltd Code of Practice: For the Manufacture, Distribution and Use of AQUI-S. Information and Guidelines for AQUI-S Use: 11.


Davies-Colley, R. J. (2008). "All cover is not created equal".


Rowe, D., M. Hicks, et al. (2000). "Reduced abundance of banded kokopu (Galaxias fasciatus) and other native fish in turbid rivers of the North Island of New Zealand." New Zealand Journal of Marine and Freshwater Research 34(3): 547-556.


