Restoring giant kokopu (*Galaxias argenteus*) populations in Hamilton's urban streams

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Abstract

In this study, options for restoring fish populations in Hamilton City (37.47ºS, 175.19ºE) were explored. Habitat and fish populations in Hamilton urban streams were manipulated using a two-fold experimental design. Firstly, habitat was enhanced in ten urban streams with three continuous treatments in a 60-m reach at each site (20 m with 10 ponga logs, 20 m with 20 hollow clay pipes, and 20 m with no added structure). Secondly, juvenile farm-reared giant kokopu (*Galaxias argenteus*), were stocked into five of the enhanced stream sections. Giant kokopu are threatened and occur naturally in Hamilton urban streams in sparse populations. The abundance of wild fish was monitored before and after enhancement and fish release from November 2006 to November 2007. Stocked fish were monitored for eight months, from April to November 2007. Over this time electric fishing was conducted three times, trap nets (Gee minnow and fyke nets) were set monthly and spotlighting was conducted monthly at three release sites where water clarity allowed.

Anticipated outcomes of this research were; to determine whether giant kokopu abundance in Hamilton urban streams is limited by recruitment or by habitat, and to assist with the development of methods to restore fish populations in Hamilton City urban streams.

Logs used as enhancement structures in Hamilton urban streams provided more stable habitat for fish and created more suitable microhabitat than pipe structures. Pipes moved considerably during high flows, and their instability made them less effective at providing habitat. Within the study sites there appeared to be complex interactions with turbidity, stream width and depth, which complicated the effect of the habitat structures. The limited replication and variability among sites contributed to statistically insignificant results using analysis of variance.

Retention and recapture rates of stocked juvenile giant kokopu were greatest at Site M11, where the stream was narrow, shallow, clear and had lower numbers and biomass of shortfin eels, compared to other survey sites. Marked and released giant kokopu were retained in the release reaches at four of the five sites, for a minimum of four months, and exhibited substantial growth. Daily growth of juvenile giant kokopu ranged from 0.19 to 0.33 mm day^{-1} and from 0.03 to 0.11 g day^{-1}, exhibiting substantial growth over winter. Giant kokopu appeared to have a slight bias to the log section of enhanced habitat, but habitat selection appeared to be overwhelmingly controlled by initial habitat selection.

The stocking of farm-reared fish into urban streams was largely successful, but the success of the habitat enhancement was variable and further work is required to determine better techniques for habitat enhancement in these urban environments. It is concluded that releasing farm–reared giant kokopu can be used to restore populations especially where recruitment limitations control fish abundance and diversity.

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Chapter 1 General Introduction

1.1 Effects of urban development

The effects of urbanisation on streams are well studied (Paul and Meyer, 2001; Morgan and Cushman, 2005; Bernhardt and Palmer, 2007). Urbanisation leads to altered hydrologic regimes, elevated nutrient and contaminant concentrations, increases in water temperature from asphalt runoff and degraded biota characterised by a few tolerant species. Morgan and Cushman (2005) describe this as the 'urban stream syndrome'. Flow in urbanised catchments tends to peak quickly with lower base flows than unaltered catchments, due to storm-water discharge from impervious surfaces; this can amplify the incidence of erosion (Bernhardt and Palmer, 2007). Storm flows can negatively affect species residing or spawning in scour-prone habitats, such as pools. Engineering designs for urban stream geomorphology often simplify channels by inhibiting natural meandering with rip-rap along stream edges, and in extreme cases they can confine streams to concrete channels or contain them in underground pipes (Bernhardt and Palmer, 2007). Urban stream environments may be described as generic because although the geology, climate and vegetation may be different, urban streams are overwhelming controlled by their urbanised impacts (Paul and Meyer, 2001).

As the world population increases into low-lying areas, the streams and rivers in these areas are becoming increasing degraded by urbanisation (Bernhardt and Palmer, 2007). High population density in urbanised areas may offer an opportunity for education, local community encounter and involvement, which is important for restoration (Paul and Meyer, 2001). Human activities in urban streams often result in elevated levels of nutrients, such as phosphorous, heavy metals from industrial and automobile discharges, pesticides and other organic contaminants (Paul and Meyer, 2001). This is likely to be the case in Hamilton urban streams but chemical characteristics were not studied here in depth as they did not fall within the scope of this thesis.

In addition to chemical changes, riparian vegetation removal can result in changes to stream morphology, causing; stream bank erosion, decreases in riffle frequency and sediment size, and an increase in suspended sediment loads (Collier et al., 1995). As a result urbanisation can lead to stream channelisation, an increase in velocity and a decrease in the amount of large wood, which can make habitat unsuitable for some species (Paul and Meyer, 2001). This may shift the fish assemblage to be dominated by pollution tolerant and habitat generalist species (Paul and Meyer, 2001; Roy et al., 2005). Wang et al. (2001) found that the amount of connected impervious surfaces (i.e. paved surfaces) in the watershed appeared to be the greatest indicator of urbanisation impacts on stream fish communities in small south eastern Wisconsin streams, USA. Paul and Meyer (2001) noted a change in fish community with increasing urbanisation where, from 0-5% urbanisation sensitive fish species are lost, 5-15% urbanisation results in habitat degradation and loss of functional feeding groups and >15% urbanisation results in high toxicity and organic enrichment severely degrading fish fauna. Urbanisation led to a restructuring in the fish community and faster growth of salmonids than in forested streams in Kelsey Creek, Washington (Scott et al., 1986). However, impervious cover and urbanisation do not give an indication of the habitat quality available, which is important for fish communities (Wang et al., 2001).

1.2 Fish habitat enhancement

Stream enhancement involves an attempt, permanent or temporary, to overcome current limitations to the ecological potential of a stream. In-stream restoration is often aimed at rehabilitating habitat that is limited by the lack of physical structure and vegetation cover. Addition of structure can improve habitat diversity, and therefore abundance of target fish species in streams (House and Boehne, 1986; De Jong et al., 1997; Crook and Robertson, 1999; Bonnett et al., 2002).

A number of factors need to be considered prior to undertaking in-stream restoration measures. Stream physio-chemical parameters should be assessed including; temperature, dissolved oxygen, pH, water depth, active width, water volume, substrate type, in-stream and riparian cover, mean velocity, bank erosion, and the number of pools before and after treatment (House and Boehne 1985; Piegay et al., 2000). In urban steams, flow alterations as a result of urbanisation also need to be considered prior to enhancement to ensure retention (Cassin et al., 2004; Herricks and Schwartz, 2004). All of these environmental factors can reduce habitat suitability for aquatic fauna, including fish.

Potential fish habitat is a function of the distribution of large woody debris, boulders, undercut banks, gravel substrate, and pools in streams (Piegay et al., 2000). Large wood has been used successfully to provide habitat for fish species in stream environments (House and Boehne, 1986; Crook and Robertson, 1999; Bonnett et al., 2002). In-stream structures can change flow dynamics, which can improve habitat suitability for some species. For example, the addition of wood in a forested stream in Germany resulted in increased water depth and a slight reduction in mean flow velocity (Mutz, 2004). Bonnett et al. (2002) artificially restored in-stream habitat for giant kokopu in small forested streams in Westland, New Zealand, using woody debris held in place with metal fencing standards, to provide cover and slow the velocity of the water.

Coupled with habitat enhancement, Piegay et al. (2000) recommend long-term and large scale perspectives, and an abandonment of the practice of removing in-stream debris, which is common practice in Hamilton urban streams to reduce flood risk. Monitoring the success of structural enhancement is imperative as the habitat enhancement practices that failed, did so shortly after installation (Brown, 2000). However, the true measure of restoration success is the response of aquatic communities (Brown, 2000).

The high ecological value of the small gully systems within Hamilton City has led to potential measures to restore and protect the indigenous vegetation of Hamilton City (Clarkson and McQueen, 2004). While vegetation has been extensively studied, there has been little work done on the urban gully system streams, which this study aims to address.

Hamilton urban streams appear to be limited by suitable in-stream habitat for aquatic organisms, in particular fish (Aldridge and Hicks, 2006). This study attempted to overcome limitations through in-stream habitat enhancement. Ponga logs and hollow clay pipes were used to restore in-stream habitat during this research. For the purpose of this study enhancement and restoration have been used interchangeably.

1.3 New Zealand fish fauna

Recent studies indicate that there are 22 introduced, and 35 native, freshwater fish species in New Zealand, many of them endemic and diadromous (McDowall, 2000). There are six families of native fish, with Galaxiidae (including the genera *Galaxias*) being the most diverse group.

Diadromy is a life history trait associated with fish, that involves often significant migration between freshwater and the ocean to complete a life cycle. Five of the native diadromous fish are *Galaxias* species which comprise the highly valued whitebait fishery in New Zealand. They are inanga (*Galaxias maculatus*) which make up the greatest proportion of the whitebait fishery on the Waikato River (Chapman, 1996), banded kokopu (*Galaxias fasciatus*), koaro (*Galaxias brevipinnis*), shortjaw kokopu (*Galaxias postvectis*) and giant kokopu (*Galaxias argenteus*). Whitebait species are amphidromous, spending time in the ocean as well as in freshwater where they grow to adulthood, although migration is not obligatory (David et al., 2004).

The fish fauna of Hamilton urban streams has been investigated by Wilding (1998) and Aldridge and Hicks (2006), these authors found a range of native fish present, notably the threatened longfin eel and giant kokopu.

1.3.1 Giant kokopu (*Galaxias argenteus*)

G. argenteus (Gmelin, 1789) (Galaxiidae) was chosen as the stocking fish because (i) they occur naturally in Hamilton urban streams (Aldridge and Hicks, 2006), (ii) they are a threatened species (Tisdall, 1994), and (iii) farm-reared individuals could be obtained. Little is known of the life history of the giant kokopu, but there is evidence that adults spawn in autumn to early winter (early April to early June) (Jellyman, 1979; McDowall and Kelly 1999). The giant kokopu is solitary and nocturnal (Bonnett et al., 2002), preferring to reside in slow flowing pools within a home range (Whitehead et al., 2002; David and Stoffels, 2003). Fish usually reside in pools adjacent to faster flowing riffles which are used by the fish to feed on drifting invertebrates (David and Stoffels, 2003). Giant kokopu have been described as generalist feeders utilising both aquatic and terrestrial food sources. Terrestrial food sources were a significant food component of the gut analysis undertaken on giant kokopu (Bonnett and Lambert, 2002).

Giant kokopu are often found in higher numbers in streams with riparian vegetation and in-stream cover (Bonnett et al., 2002). The five habitat features found to be associated with the occurrence of giant kokopu were (in order of importance); in-stream cover, water depth, low water velocity, proximity to the ocean, and shade/riparian cover (Bonnett and Sykes, 2002). Baker and Smith (2007) found that giant kokopu strongly selected debris dams and undercut banks as habitats in small streams draining the Hakarimata Ranges, New Zealand.

The decline of the giant kokopu has been linked to loss of habitat and increased competition from introduced salmonids (David, 2002). Larger galaxiids and eels are likely to inhabit deeply undercut banks and are often found associated with pools, in-stream cover and low water velocities (Bonnet et al., 2002; David, 2002; Richardson and Taylor, 2002).

Suitable habitat is needed for the conservation of this species (Minns, 1990; Bonnett and Sykes, 2002). Therefore, in-stream habitat restoration by adding stream complexity and changing local stream morphology to form pools might be expected to increase giant kokopu survival, abundance, and biomass.

1.4 Thesis aims & objectives

The overall aim of this study was to develop tools to restore the biodiversity of fish in urban streams, by addressing three objectives:

1. To determine the response of wild fish to the addition of habitat enhancement structures,
2. To determine whether farm-reared juvenile giant kokopu would survive, grow and remain in urban streams, in particular release reaches, and
3. To investigate whether fish populations in Hamilton City urban streams were limited by habitat or recruitment.

I hypothesise that enhancing urban stream habitat through the addition of in-stream structure will add habitat for giant kokopu, thereby increasing their abundance in the restored sections of stream. Urban streams are likely to be habitat limited due to clearing of riparian vegetation, storm-water effects and physical removal of in-stream structures which is routinely practiced in Hamilton City urban streams. Using two different structures enabled two potential enhancement measures to be compared.

Objective one was addressed by extensively fishing urban streams prior to habitat addition to determine the existing fish populations at each of the 10 sites. Clay pipes and ponga logs were introduced to two 20-m reaches in each stream and the third reach was left as a control. Each 20-m reach was monitored over eight months from April to November 2007 using trap nets and electric fishing to determine changes in fish population as a result of habitat addition. Five Gee minnow nets and fyke nets where water depth permitted were set overnight in each of the 20-m reaches monthly and electric fishing was conducted three times over the duration of the study. Fish were measured to determine changes in density and biomass as a result of structural addition.

Objective two was addressed by releasing marked farm-reared juvenile giant kokopu to five of the 10 restoration sites. Populations were monitored monthly with trap nets at all sites and at three sites by spotlighting. All sites were electric fished three times throughout the study. Fishing methods were compared to determine the method most effective at capturing farm-reared juvenile giant kokopu, and caught fish were measured to determine growth of fish within the streams. The relationship between released giant kokopu and wild fish populations was explored.

The third objective of this study was to determine whether fish populations in Hamilton urban streams were recruitment or habitat limited, arising from access to streams or a lack of habitat, respectively. The limitations on streams will be determined by assessing the habitat selection by wild fish and the survival and retention of released juvenile giant kokopu.

Hamilton City is currently experiencing growth in urbanisation with several new housing and amenity developments proposed within the next 10 years. It is hoped that habitat enhancement and stocking of a farm-reared threatened species has the potential to be used for restoring the biodiversity of urban streams and giant kokopu populations. The results of this research will be used as a tool by managers, when restoring and managing the ecology of the urban streams within Hamilton City (e.g., Hamilton City Council and Environment Waikato for Hamilton streams).

1.4.1 Structure of Thesis

The thesis is divided into four sections; Chapter two will discuss background and historical information of the research area including a description of Hamilton City streams, New Zealand, where the study took place. Chapter three will discuss fishing methods used to capture fish, and the response of wild fish populations to in-stream habitat enhancement. Chapter four will outline the growth of released giant kokopu, their response to in-stream habitat structures, and to other fish, in particular the presence of shortfin eels and 'wild' giant kokopu. The last chapter of this thesis involves an integrated summary of the results presented and explained in chapters three and four. This chapter includes recommendations for enhancing fish populations in urban streams and further restoration work.

Chapter 2 Study Area

2.1 General setting

Hamilton City is located in the Waikato Region of the North Island, New Zealand (Figure 1). Underlying geology comprises the Hinuera Surface, an alluvial fan composed of gravel and sand, formed by the wandering movement of the Waikato River (McCraw, 2002). About 14, 000 years ago, with less volcanic activity within the Taupo Volcanic Zone, the Waikato River became confined to its present course. Hamilton City is bisected by the Waikato River, which originates from Lake Taupo and meanders for 425 km through the Waikato basin before flowing to the ocean at Port Waikato. The Waikato River is the longest river in New Zealand and has a catchment area of 14, 258 km^2 (Chapman, 1996).

Several discharges enter the Waikato River south of Hamilton City. The 5-year median data from a Waikato River water quality monitoring programme for 2007 revealed that many water quality parameters became more degraded from upstream to downstream (Beard, 2007a). The monitoring programme is based on data from several stations along the river including south of Hamilton at Narrows, and north of Hamilton at Horotiu. Biological oxygen demand is a measure of bacterial activity and increases along the river length. Faecal coliforms increase from 50/100 mL at Narrows to 120/100 mL in Horotiu, an indication that faecal matter enters the river via streams in Hamilton City. Turbidity and nutrient levels increase along the river reach indicating enrichment (Beard, 2007a).

2.1.1 Hamilton City

Hamilton City (37.47ºS, 175.19ºE) is New Zealand’s seventh most populated city covering a highly modified area of 9, 427 ha (Clarkson and McQueen, 2004). Gully streams within the city are an important natural feature and were formed from the erosion action of spring sapping, which causes slips, thereby creating a network of streams draining into the Waikato River (McCraw, 2000). The repetition of this process has lead to the formation of the steep-sided gully systems and streams which comprise approximately 750 ha or 8% of Hamilton City urban area (McCraw, 2000; Clarkson and McQueen, 2004). There are four major catchments located within Hamilton City; Waitawhirwhiri, Mangaonua, Kirikiriroa and Mangakotukutuku all of which have headwaters originating in farmland outside the city boundary (Wall and Clarkson, 2001). Prior to draining for pasture, peat was extensive in this area, and as a result many of the streams within the city have naturally peat-stained water. The Mangakotukutuku and Kirikiriroa catchments are the largest in the city. The Waitawhiriwhiri catchment has limited fish access caused by the presence of a perched culvert at the confluence with the Waikato River, but fish access to other major catchments is generally good (Aldridge and Hicks, 2006).

The lowland nature of the area is reflected in the characteristics of the streams, water quality and organisms that inhabit them. Stream-beds are typically dominated by fine sediments, partly reflecting low water velocities, and channels have a mixture of runs, pools and backwaters with extensive riffle areas uncommon; however, small riffles were present at sites K10, K10b, K11, K2, M1 and P1 (see below).

2.2 Selection of study sites

Ten sites within Hamilton were selected for this study (Figure 1; Plates 1-10). All 10 sites except one (M12) were located in public parks managed by the Hamilton City Council. Sites were chosen because they were (i) representative of urban streams in Hamilton, (ii) relatively easy to access but were far enough away from public view such that structural tampering would be minimal, and (iii) had homogeneous 60-m stretches where most physical environmental variables appeared constant (e.g. no tributary inputs, storm-water discharges). The 10 study sites were located within three catchments; five sites were located in the Kirikiriroa (K) catchment in north Hamilton (Figure 1; Plates 1-5). Three of these were within the main branches of three conjoining streams within Mangaiti Park (Table 3). The other two sites were Chartwell Park (K11) and Tauhara Park (K2), and were on the same stream but several hundred metres apart. Four of the 10 study sites were located in the Mangakotukutuku (M) catchment (Figure 1; Plates

6-9) on four separate arms of the main stream. Two of the sites were located in Sandford Park, while the other two were further upstream in Te Anau Park and at the Melville Marae on Collins Road. The last site was situated in the centre of the city in Parana Park (P; Figure 1; Plate 10). The Kirikiriroa and Mangakotukutuku catchments have headwaters that begin in pasture before travelling through the city to the Waikato River, whereas the Parana Park catchment is entirely urbanised. All sites were used for habitat enhancement (Chapter 3), and five of these sites were randomly selected for introduction of giant kokopu (see Chapter 4).

All sites were partially shaded by the surrounding riparian vegetation, which comprised a mixture of native and introduced species. Most of the sites had low growing introduced weeds or grasses lining stream banks. The exception was Parana Park (P1) where the stream was a feature, and as such had maintained, predominantly native plantings along its banks. Site M7, in Te Anau Park had a top canopy dominated by eucalyptus trees (*Eucalyptus* sp.) with an understorey of wheki (*Dicksonia squarrosa*) and wandering willie (*Tradescantia fluminensis*). Wandering willie was the dominant species along the stream bank at sites M1 and K2, and was common at sites K11 and M12. Lichens were observed growing in the seeps along the stream bank at Site K11. Site K10 had an established native tree upper storey, while K10a and K10b, also located in Mangaiti Park, had recently planted native trees (dominated by cabbage tree (*Cordyline australis*) and flax (*Phormium tenax*)). Several large crack willows (*Salix fragilis*) provided in-stream shade at Site K10a. Rank grasses and leaf litter provided ground cover under a wheki-dominated canopy at Site M11.

Prior to habitat enhancement, substrate at most sites was dominated by fine sediment except at K2 where gravel dominated, and M12 where cobble to boulder-sized sediment was common. Cobble/boulder sediment at site M12, and to a lesser extent K11, K2 and P1, were artificial introduced to the streams to provide a habitat feature (P1, K11 and K2) and to stabilise banks (M12).

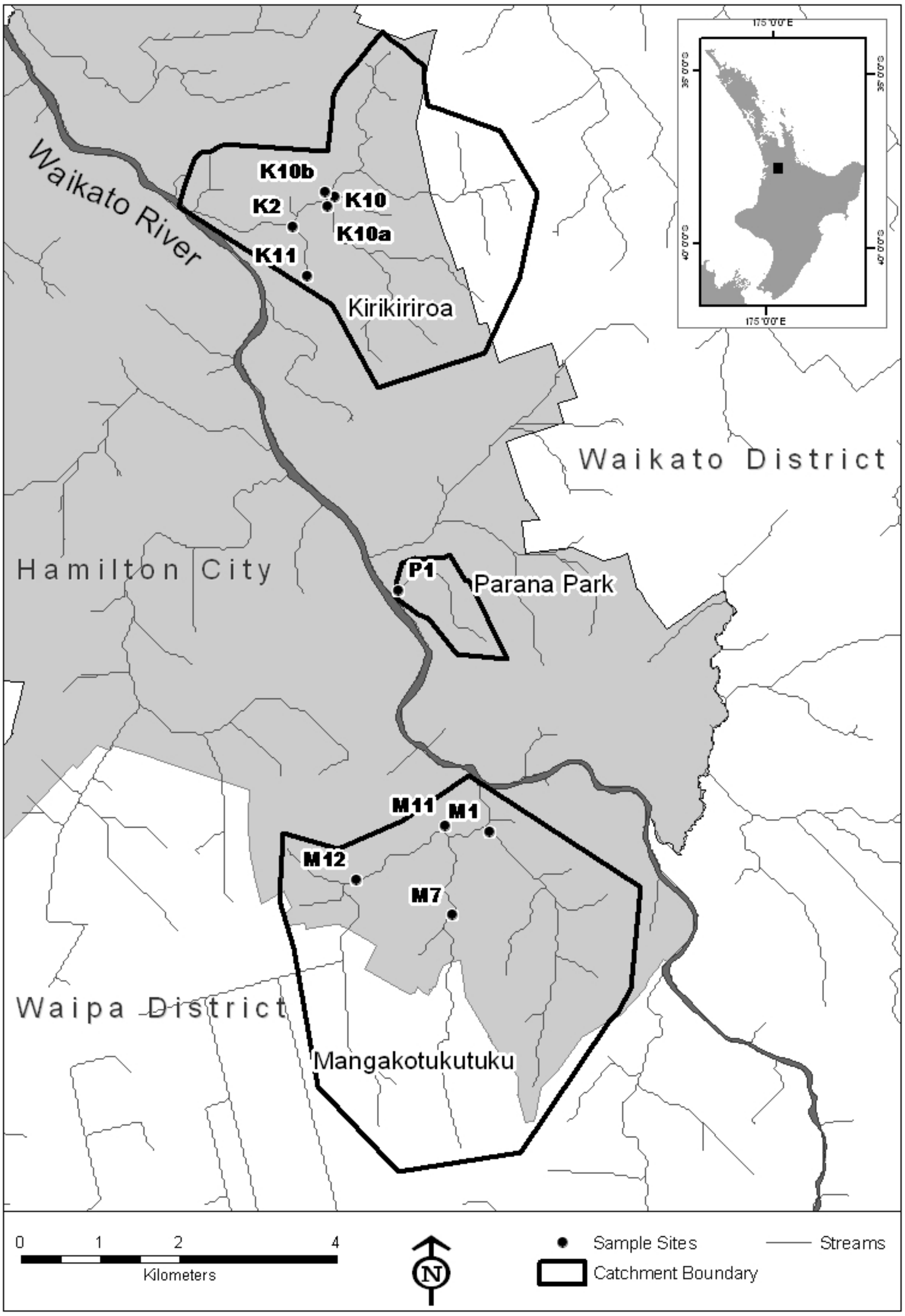

Figure 1: Sample sites located with Hamilton City urban area (grey). Five sites are located within the Kirikiriroa catchment (K10a, K10b, K10, K11, K2). One site is located in the middle of the city in close proximity to the Waikato River (P1), and the remaining four sites are located in the Mangakotukutuku catchment (M1, M7, M11, M12) at the southern end of Hamilton. (Territorial Authority Boundaries obtained from Statistics New Zealand data).

Plate 1: Site K2 located in Tauhara Park in the Kirikiriroa catchment. Photo has been taken looking downstream from within the log treatment section.
NZ grid ref: E 2710250, N 6381720

Plate 2: Site K10 located in Mangaiti Park within the Kirikiriroa catchment. Photo has been taken looking upstream from the control section.
NZ grid ref: E 2710753, N 6382107

Plate 3: Site K10a located in Mangaiti Park within the Kirikiriroa catchment. Photo is taken looking upstream from the bottom of the log section.
NZ grid ref: E 2710728,
N 6381957

Plate 4: Site K10b located in Mangaiti Park within the Kirikiriroa catchment. Photo has been taken in the log section looking downstream.
NZ grid ref: E 2710729, N 6382160

Plate 5: K11 located within Chartwell Park on Bankwood Road in the Kirikiriroa catchment. Photo has been taken looking downstream in the log treatment section.
NZ grid ref: E 2710435, N 6381106

Plate 6: M1 located in Sandford Park within the Mangakotukutuku catchment. Photo has been taken looking upstream from the log treatment section of the stream.
NZ grid ref: E 2712751, N 6374066

Plate 7: Site M7 located in Te Anau Park within the Mangakotukutuku catchment. Photo shows the log section looking upstream.
NZ grid ref: E 2712278, N 6373010

Plate 8: M11 located in Sandford Park within the Mangakotukutuku catchment. Photo has been taken looking downstream in the pipe treatment section.

NZ grid ref: E 2712189, N 6374138

Plate 9: Site M12 located at the Marae on Collins Rd, Melville, upstream of Ohaupo Rd culvert. Photo taken looking upstream from the log section.

NZ grid ref: E 2711063,
N 6373458

Plate 10: Site P1 located in Parana Park approximately 50 m from the Waikato River. Photo has been taken from the top of the log section looking downstream toward the Waikato River.

NZ grid ref: E 2711696, N 6376986

2.3 Physico-chemical conditions

Water quality data are collected monthly by the Waikato Regional Council (Beard, 2007b) within the lower parts of the Kirikiriroa and Mangakotukutuku catchments, both of which were studied during this research. The five year median values from this sampling regime are displayed in Table 1 and are discussed in relation to guidelines for the water quality data outlined by Beard (2007b). The dissolved oxygen levels for Mangakotukutuku Stream were excellent; however, the results for Kirikiriroa Stream were below the satisfactory level for maintaining aquatic life. The nutrient levels of the stream water were higher in the Mangakotukutuku Stream than the Kirikiriroa Stream, probably due to farming practices in the headwaters of the Mangakotukutuku catchment, which has a greater rural influence. The total nitrogen and phosphorus levels, which can result in nuisance plant growth, were elevated above satisfactory levels in both catchments. Ammonical nitrogen is toxic to aquatic life in high amounts, and results were satisfactory within both catchments (Beard, 2007b). The pH of the streams was satisfactory and excellent within the Kirikiriroa and Mangakotukutuku catchments, respectively. Turbidity and *Escherichia coli* levels were elevated far above satisfactory levels for human recreational use in both catchments.

Table 1: Water quality parameters for the Kirikiriroa and Mangakotukutuku catchments collected above the confluence with the Waikato River at Tauhara Drive and Peacockes Road, respectively. Results displayed are median values calculated from monthly data from 2002 to 2006. Satisfactory and excellent thresholds are based on guidelines used by Beard (2007b).

Water Quality parameters	Units	Kirikiriroa	Mangakotukutuku	Satisfactory	Excellent
Dissolved oxygen	%	69	90	>80	>90
Conductivity @ 25ºC	µS cm^{-1}	209	189		
Dissolved reactive phosphorous (DRP)	g m^{-3}	0.02	0.14		
Total phosphorus (TP)	g m^{-3}	0.09	0.42	<0.04	<0.01
Ammonical nitrogen (NH_4N)	g m^{-3}	0.33	0.31	<0.88	<0.1
Nitrate/nitrite (NOxN)	g m^{-3}	1.06	1.08		
Total Kjedahl nitrogen (TKN)	g m^{-3}	0.84	1.39		
Total nitrogen (TN)	g m^{-3}	1.99	2.42	<0.5	<0.1
Turbidity	NTU	30	25	<5	<2
pH		6.9	7.0	6.5-9	7-8
E. coli	no. 100 mL^{-1}	475	1450	<126	<23
Enterococci	no. 100 mL^{-1}	550	685		
Temperature	ºC	15	15		

Selected physical and water quality parameters were measured monthly in the present study when trap nets were set at each of the sites. Results are presented in this section because they were not expected to change with fish habitat enhancement (Table 2). Average stream width was 1.5 m with M12 being the widest stream at 3.48 m and P1 and K10a exceeding 2 m. M12 was also the deepest stream with an average depth of 0.56 m, followed by K10a (0.35 m) and K10b (0.30 m).

Temperature within these small urban streams varied widely among sites over time, from 5.3°C (minimum in winter at K10, a small shaded stream) to 20.6°C (in summer at K10) (Table 2). The standard error of the temperature within a site was low and within the range 1.3 to 3.7°C, with P1 having the smallest temperature range. This stream also had the highest average temperature and a large amount of storm-water discharge; here shade may act to keep artificially elevated in-stream temperatures relatively stable. Other water quality parameters that were measured during the study were dissolved oxygen, conductivity and pH.

Dissolved oxygen had an average saturation of 77%, with a range between 35% and 98% (Table 2). The lowest reading was taken at M7 in January 2007 after a thick microbial growth had established on rock surfaces. This growth was quickly washed out of the stream by the next large rainfall event. The average conductivity reading taken was 181 µS cm^{-1} ranging from 127 to 375 µS cm^{-1}. Conductivity was greatest and had more variability at sites K10 and M7. The pH of the streams was similar and circum-neutral, varying from 6.4 to 7.8 with an average of 6.6 over all sites (Table 2). Turbidity, assessed visually on a five point scale, was “clear” to “slightly turbid” at seven of the ten sites. Turbidity was “high” at sites K10a and M12, and peat stained at M7.

An overall qualitative assessment of habitat quality was conducted based on parameters that score riparian, bank and channel habitat on a scale of one to 20. This habitat quality assessment was based on the protocol from Environment Waikato ‘REMS’ monitoring guidelines (Collier and Kelly, 2005) and has a maximum score of 180. Habitat scores at the urban sites ranged from 80 to 116, and were greatest at sites K10, M1 and M7. K10a was considered to have the poorest habitat quality of the streams studied (Table 2).

Table 2: Average (± 1 Standard error) water quality results collected on one date prior to habitat enhancement and then monthly over the duration of the study. The average and range of results are displayed for each of the water quality parameters. Habitat quality scores are derived from the qualitative assessment of nine variables (see Collier and Kelly, 2005), and has a maximum of 180.

Site	Location description	Stream width (m)	Stream depth (m)	Temperature (°C)	Dissolved oxygen (%)	Dissolved oxygen (mg L^{-1})	Conductivity (µS cm^{-1} @ 25°C)	pH	Turbidity	Habitat quality
Kirikiriroa catchment										
K2	Tauhara Park	0.97 ± 0.07	0.14 ± 0.03	13.5 ± 2.8	84 ± 2.7	8.89 ± 0.69	181 ± 10	7.1 ± 0.29	clear	100 ± 1.6
K10	Mangaiti Park	1.07 ± 0.1	0.14 ± 0.02	12.6 ± 3.7	76 ± 5.5	8.18 ± 1.05	211 ± 64	7.1 ± 0.24	clear	112 ± 1.6
K10a	Mangaiti Park	2.49 ± 0.27	0.35 ±0.01	14.4 ± 1.7	72 ± 7.1	7.35 ± 0.76	180 ± 20	7.2 ± 0.24	high	97 ± 1.0
K10b	Mangaiti Park	1.00 ± 0.07	0.30 ± 0.01	12.8 ± 2.2	81 ± 7.0	8.60 ± 0.72	204 ± 30	7.0 ± 0.29	slight	113 ± 1.8
K11	Chartwell Park	1.21 ± 0.22	0.14 ± 0.06	14.8 ± 2.0	83 ± 6.2	8.46 ± 0.86	194 ± 13	7.1 ±0.32	clear	102 ± 3.0
Mangakotukutuku catchment										
M1	Sandford Park	1.42 ± 0.02	0.13 ± 0.00	12.1 ± 2.7	90 ± 4.3	9.73 ± 0.83	189 ± 18	7.3 ± 0.20	clear	112 ± 2.1
M7	Te Anau Park	1.34 ± 0.34	0.16 ± 0.04	13.8 ± 2.4	86 ± 17.2	9.04 ± 2.02	200 ± 64	7.21 ± 0.26	stained	113 ± 3.2
M11	Sandford Park	1.10 ± 0.64	0.18 ± 0 .00	12.2 ± 2.1	84 ± 5.5	9.00 ± 0.74	202± 18	7.2 ± 0.30	slight	110 ± 3.1
M12	Melville Marae	3.48 ± 0.12	0.56 ± 0.06	13.5 ± 2.3	82 ± 5.0	8.50 ± 0.74	201 ± 16	6.8 ± 0.37	high	102 ± 1.5
Gibbons Creek										
P1	Parana Park	2.13 ± 0.14	0.21 ± 0.02	15.9 ± 1.3	92 ± 4.0	9.13 ± 0.33	181 ± 21	7.2 ± 0.10	slight	101 ± 2.5
Average		1.5	0.2	12.7	77.3	8.1	180.6	6.6	clear	107
Range		0.9-3.5	0.1-0.6	6.5-20.6	35-98	3.3-11.2	127-375	6.4-7.8	clear - high/stained	80-116

2.4 Fish populations

There are nineteen species of native fish within the Waikato River, comprising three marine wanderers and ten species of introduced fish (Table 3). Thirteen fish species occur in Hamilton urban streams (* in Table 3) including the threatened longfin eel and giant kokopu (Aldridge and Hicks, 2006). The 13 species include eight native and five introduced species. Of the introduced species three are considered to be pest fish; mosquitofish (*Gambusia affinis*), catfish (*Ameriurus nebulosus*) and koi carp (*Cyprinus carpio*). The habitat occupied by these pest fish, as well as the effect they have on the environment, differs between species. Mosquitofish prefer to occupy slow flowing water such as wetlands and margins where aquatic plants grow. These fish are able to tolerate a wide range of adverse water conditions, and breed prolifically where conditions are ideal. Catfish occupy still waters such as weed-choked streams and lakes. Koi carp also prefer slow, weedy streams, rivers, and lakes. Koi carp are robust and can survive out of water if kept moist. Both catfish and koi carp, feed along the bottom of water-ways and can increase water turbidity making habitat unsuitable for other fish species (McDowall, 2000).

Forty one sites were fished in Hamilton City by Aldridge and Hicks (2006) using a combination of trap nets, electric fishing and spotlighting. Shortfin eels (*Anguilla australis*) were the most common species encountered by site, being caught or observed at 23 of the sites fished. Mosquitofish were the most numerous species captured, with 270 individuals captured in trap nets in a stream in the Rotokauri catchment. The two main catchments sampled were the Kirikiriroa and Mangakotukutuku catchments with 10 sites each. Longfin eels were more common in the Mangakotukutuku catchment. Giant kokopu were caught at three sites, two sites in the Kirikiriroa catchment and one in the Mangakotukutuku catchment. Catfish and torrentfish were captured at one site each, while trout and koi carp were captured at two sites. These fish were considered to be uncommon within the city streams, and it appeared that koi carp absence may be maintained by culverts preventing passage at some sites.

Table 3: Fish found within the Waikato River (source: Waikato Regional Council/EW website). (* = recorded in Hamilton urban streams by Aldridge and Hicks, 2006).

Native Fish		Introduced Fish	
Common Name	Scientific Name	Common Name	Scientific Name
Yellow-eyed mullet	*Aldrichetta forsteri*	Catfish *	*Ameiurus nebulosus*
Shortfin eel *	*Anguilla australis*	Goldfish	*Carassius auratus*
Longfin eel *	*Anguilla dieffenbachii*	Grass carp	*Ctenopharyngodon idella*
Australian longfin eel	*Anguilla reinhardtii*	Koi carp *	*Cyprinus carpio*
Lamprey	*Geotria australis*	Mosquitofish *	*Gambusia affinis*
Torrentfish *	*Cheimarrichthys fosteri*	Rainbow trout *	*Onchorhycchus mykiss*
Giant kokopu *	*Galaxias argenteus*	Perch	*Perca fluviatilis*
Koaro	*Galaxias brevipinnis*	Brown trout *	*Salmo trutta*
Banded kokopu *	*Galaxias fasciatus*	Rudd	*Scardinius erythropthalmus*
Inanga *	*Galaxias maculatus*	Tench	*Tinca tinca*
Short-jawed kokopu	*Galaxias postvectis*		
Black mudfish	*Neochanna diversus*		
Giant bully	*Gobiomorphus gobiodes*		
Common bully *	*Gobiomorphus cotidianus*		
Redfin bully	*Gobiomorphus huttoni*		
Cran's bully	*Gobiomorphus basalis*		
Grey mullet	*Mugil cephalus*		
Common smelt *	*Retropinna retropinna*		
Black flounder	*Rhombosolea retiaria*		

Chapter 3 Response of fish to in-stream habitat enhancement

3.1 Introduction

It is widely recognised that urban streams in general, are limited in their biological potential by hydrological conditions and contaminants delivered from upstream impervious surfaces (Paul and Meyer, 2001; Wang et al., 2001; Herricks and Schwartz, 2004; Roy et al., 2005). These streams usually have elevated contaminant and nutrient concentrations, such as heavy metals and phosphorus; streams may also be influenced by waste water and nonpoint effluent (Paul and Meyer, 2001). Flashy flows are characterised by sediment and structural movement as well as displacement of biota. These changes, which are coupled with human population density, are termed the ‘urban stream syndrome’ (Morgan and Cushman, 2005). Urban catchments are known to alter storm responses resulting in flashier hydrographs, a faster onset of storm flows and earlier and higher peak discharges in streams (Farahmand et al., 2007). Peak flows enter streams more rapidly due to less infiltration, which increases erosion, creating channelisation, higher stream velocities, and changing stream geomorphology (Paul and Meyer, 2001). Wang et al. (2001) stated that greater flooding leads to a loss of pool habitat and in-stream cover, emphasising the importance of restoring cover to provide refugia for fish in urban streams. Survey work by Aldridge and Hicks (2006) demonstrated that some fish species notably, shortfin and longfin eel, smelt, banded kokopu, inanga and giant kokopu, can survive in Hamilton urban streams where habitat conditions provided refuge from high flows.

Stable physical structure in streams, e.g, in the form of tree roots or large wood, is important as it can provide refugia for fish from high flows and predation, can decrease erosion, and is an area for food production. Stream restoration initiatives in the past have addressed stream functioning and habitat limitations caused by a lack of physical structure, where the type of restoration undertaken depends on the limitation being addressed. Riparian vegetation along stream banks is often replanted to decrease in-stream temperatures and encourage normal stream functioning through nutrient, allochthonous organic matter and large wood input (Wissmar and Beschta, 1998; Parkyn et al., 2005). Large wood is often used to restore habitats in low velocity streams and rivers (House and Boehne, 1986; Bonnett and Sykes, 2002; Kail and Hering, 2005; Cordova et al., 2007), while artificially placed boulders can enhance habitat in faster flowing waters (Negishi and Richardson, 2003). Cordova et al. (2007) found that large wood was often the only significant stable substrate in Midwestern streams, and was therefore an important ecological feature. Large wood is usually more common in pools, particularly in streams surrounded by older established forests (Evans et al., 1993).

Habitat restoration has been used as a tool to restore fish populations, and often involves the addition of structures that can improve general habitat. Habitat provides cover and refugia, areas for spawning, complexity and increased food sources, through the development of secondary channels, pools, meanders, undercut banks and increased retention of organic matter (House and Boehne, 1986; Cordova et al., 2007). House and Boehne (1985) found that coho salmon (*Oncorhynchus kisutch*) and steelhead (*Oncorhynchus mykiss*) spawning increased substantially in habitat artificially restored using ‘v’ shaped gabian baskets, boulders and logs in a coastal Oregon stream. Artificial riffle structures have been used to restore habitats of the threatened Neosho madtom (*Noturus placidus*) fish with success in Kansas (Fuselier and Edds, 1995).

Few published studies have investigated factors affecting fish distribution in New Zealand urban streams. The fish species assemblage of Hamilton urban streams is dominated by native species, with shortfin eels being the most abundant (Wilding, 1998; Aldridge and Hicks, 2006) (see Chapter 2). Wilding (1998) suggested that the high number of eels might be a function of eel’s ability to gain access to, and survive, in adverse conditions. Shortfin eels were also found to be the most common species encountered in pasture streams in the Waikato Region by Hicks and McCaughan (1997) and Hicks (2003), who analysed the fish communities at eleven sites on the Mangaotama Stream in the Waikato region. In general eel density and biomass was greater within the pasture sites than native forest sites. Jowett and Richardson (2003) compared fish communities with environmental variables, and found that shortfin eels preferred farmed over native catchments, reflecting finer substrate and possibly warm water (Richardson et al., 1994). Many of the native species found in Hamilton urban streams are diadromous, making access to the streams an important determinant of presence. Culverts within Hamilton City catchments assessed by Aldridge and Hicks (2006) mostly provided sufficient access to study sites from the Waikato River, the exception to this was Site K11 where a perched culvert existed between the site and the Waikato River.

Stream habitat was enhanced during this study using hollow clay pipes and ponga logs (*Dicksonia squarosa*); these structures were monitored for eight months to gauge the success of restoration initiatives, as recommended by Ebersole et al. (1997). Coarse particulate organic matter (CPOM) retained around structures in streams can provide a food source for macroinvertebrates which in turn can provide food for fish and refugia for fish from predation (Richardson and Niell, 1991). Wallace et al. (1997) found that CPOM was essential for conserving or restoring diverse stream food webs, with a loss of detritus resulting in a decrease in abundance and biomass of invertebrates.

Structures within a stream can influence sediment transport and storage, thereby altering microhabitat conditions. Borg et al. (2007) continually monitored scour produced by logs in a sand-bed dominated stream in the Snowy River, SE Australia using a pressure transducer. These authors found complex interactions with flow in the four pool types studied, where the plunge pools appeared to fill on the rising limb of the hydrograph and scour on the falling limb, differing from the hypothesis that scour would increase with discharge. Sediment storage was greater behind woody debris, and wood has also been found to increase microhabitat by accumulating sediment in both the scour and depositional areas of a stream (Gomi et al., 2001). Sediment transport is a function of discharge which is often elevated in urban streams (Paul and Meyer, 2001; Suren and McMurtrie, 2005).

The aims of this chapter are to:

1. Monitor the effectiveness, retention and microhabitat formation around the artificially added log and pipe structures,
2. Determine changes in fish populations before and after habitat enhancement, and
3. Explore any preference of the fish populations or individual species to habitat enhancement.

3.2 Methods

3.2.1 Addition of structures

Habitat was enhanced in 10 stream reaches within the Mangakotukutuku and Kirikiriroa catchments and Parana Park in Hamilton City (see Chapter 2). All of the streams were within Hamilton City boundaries and considered urban due to the influence of housing development, impervious cover, and storm-water discharge. Two different structures were used to restore habitat within these urban streams; circular hollow clay pipes (400 X 150 mm) and sections of ponga log (~0.8 m long by 200 mm in diameter). The structures were either attached to the stream bed or stream bank with lengths of 6 mm reinforcing bar. Reinforcing bar was hammered into the bed sediment at each end of the ponga log, and clay pipes were secured perpendicular to the stream bank with a piece of reinforcing bar through the circular opening. Structures were added to the streams over a two week period from the end of February to the beginning of March (25 February, 1 and 10 March 2007). Following the introduction of the structures, structural integrity was monitored at monthly intervals.

Sixty metre experimental sections in the 10 urban streams were chosen for this study. Each 60-m section was partitioned into 20-m reaches; two reaches were restored using in-stream habitat enhancement structures, and one reach was left untreated as a control. The allocation of treatments within each 60-m reach was control, pipe and then ponga log from an upstream to downstream direction (Figure 2). Treatments were allocated in this way so that the influence of upstream treatments on downstream stream hydraulics was minimal. Ten ponga logs were oriented at a 45º angle downstream in each 20-m log treatment section on alternate banks. Twenty hollow clay pipes were secured perpendicular to the stream bank, on alternate sides, in each 20-m pipe treatment reach. These orientations were chosen because they were considered to provide refugia for fish from high water velocities experienced during floods.

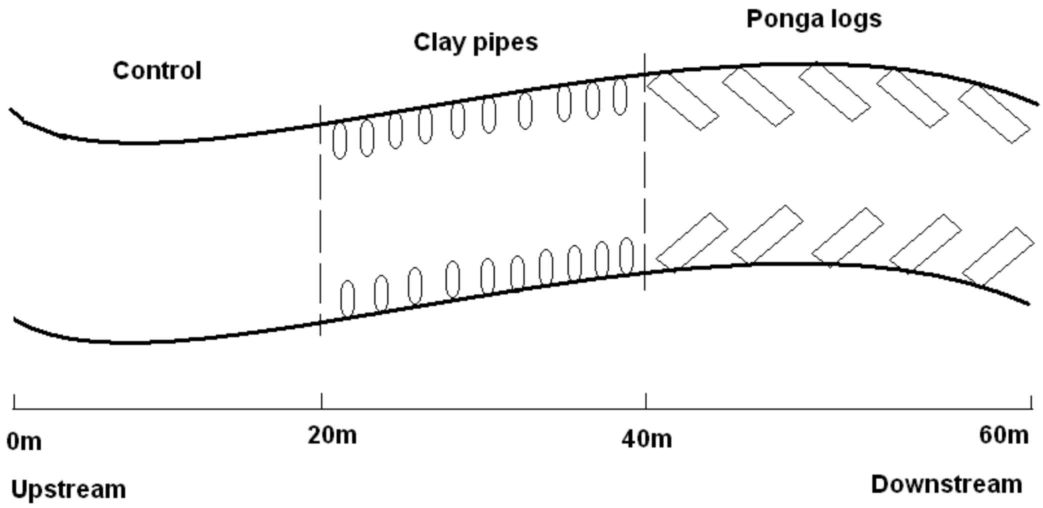

Figure 2: Schematic of the allocation of treatments to a standard section of stream. Each site consisted of; a 20-m control, a 20-m pipe treatment and a 20-m log treatment. Not drawn to scale.

3.2.2 Monitoring structural changes

The stream section in the immediate vicinity of the structures was monitored monthly for several changes, including the number of structures visible, organic matter build-up, and scour and deposition immediately upstream and downstream of the structures. Structures were assessed visually and recorded in percentile abundance classes (0, 0-25, 26-50, 51-75, >76). Visual assessments, rather than direct physical measurements, were carried out to minimise physical disturbance to the treatments. Organic matter, scour, and deposition were recorded as percentile coverage of the visible structures.

Several stream characteristics were measured monthly at each of the sites immediately prior to fish monitoring (see Table 2). These characteristics included dissolved oxygen (DO), pH, temperature, conductivity, water depth, active channel width, water volume, substrate type, in-stream and riparian cover, bottom sediment composition, and velocity. Dissolved oxygen was recorded using a hand-held YSI Model 55 DO meter, pH was measured with a hand-held waterproof pH Testr 2 (Eutech and Oakton instrument), conductivity and temperature were measured using a using a hand-held YSI Model 30 conductivity, salinity and temperature meter. All other parameters were assessed visually.

3.2.3 Fishing survey and measurement

Prior to habitat restoration all sites were extensively fished using trap nets (Gee minnow and fyke nets), and electric fishing. Initial fishing determined the species present and their abundance. Trap nets were set overnight, over a 24 hour period ± 8 hours. Electric fishing is a common fishing method used to assess fish populations in wadeable streams (Wilding, 1998; Hicks and McCaughan 1997; Hicks, 2003; Aldridge and Hicks, 2006). Trap nets were used to supplement fishing results within this study because they could be set more often with less effect on fish species and habitat. Fyke nets were used in deeper streams to capture larger fish which are more likely to reside in (deeper/wider) streams that could support a higher biomass.

Following habitat enhancement the populations of wild fish were monitored at monthly intervals using unbaited trap nets and, twice (June and November) electric fishing. Five Gee minnow nets and a single fyke net (where pools permit) were set in each of the 20-m treatment reaches. Pest fish that were caught (i.e. koi carp, rudd, catfish and mosquitofish) were removed and killed humanely in accordance with Animal Ethics Committee Standards. Electric fishing of the sites was conducted three times throughout the duration of the research to minimise disturbance to the introduced structures and associated habitat changes. Single pass electric fishing was conducted prior to habitat enhancement to determine initial fish abundance and population structure, and then in June and November, three and eight months after habitat addition, respectively. Single-pass electric fishing is known to capture about half of the estimated population (Jowett and Richardson, 1996).

The total length of all fish caught was measured to the nearest millimetre in the field using a fish measuring board. Fish were either weighed in the field using a balance or their weight was calculated using length – weight relationships in the form: $W = e^{a}L^{b}$. The equations presented in Table 4, were used to calculate the weight of fish from the length data.

Table 4: Equations used to calculate fish species weight from a given length. The formula for the equation has been adapted from regression plots for each of the species. All equations have the form $W = e^{a}L^{b}$, where W = weight and L = length.

Fish species	N	Constant (a)	Slope (b)	r^2	Adapted from
Shortfin eel	261	-13.95	3.10	0.99	Hicks and McCaughan (1997)
Longfin eel	113	-15.12	3.35	0.99	Hicks and McCaughan (1997)
Common smelt	62	-14.91	3.75	0.94	Hicks and McCaughan (1997)
Banded kokopu	56	-13.26	3.36	0.99	McCullough and Hicks (2002)
Giant kokopu	317	-13.15	3.37	0.98	David, B.; Aldridge, B.
Common bully	115	-11.71	3.10	0.98	Hicks, B. (2007 unpublished)
Torrentfish	125	-12.16	3.19	0.99	Glova et al. (1985)
Inanga	128	-13.07	3.22	0.89	Hicks, B. (unpublished)
Mosquitofish	100	-13.15	3.53	0.96	McDonald, A. (2007)
Catfish	247	-11.64	3.08	0.98	Hicks, B. (unpublished)
Rudd	294	-12.73	3.37	0.99	Hicks, B. (unpublished)

3.2.4 Statistical analysis

Average and total rainfall over the study period has been used as a surrogate for stream flow to explore structural movement as a result of high flow levels within Hamilton urban streams. Structural retention over time is displayed as a means plot in Figure 4. The number of structures that moved from the secured position to another locality within the restoration reach was recorded monthly. Movement has been defined as ‘displaced from the point of attachment’. The stability of the structures added to the stream has been compared using percentile average retention over time across all sites. Organic matter accumulation, and sediment deposition and scour around the structures is displayed using weighted percent, calculated from the midpoint of each of the five abundance classes (section 3.2.2) multiplied by the number of structures in each abundance class, and then divided by the total number of structures seen at each site. The weighted percent has been averaged over all sites to show monthly and seasonal trends, error bars are not shown for clarity.

Frequency of species caught was used to compare fish captured over fishing methods, season and habitat. Biomass and density plots (means ± 95 % confidence interval) indicate use of the enhanced reaches. The records for the stocked giant kokopu have been removed from the data and are displayed separately in Chapter 4. All native fish have been sampled with replacement and pest fish were removed and killed humanely. One-way analysis of variance (ANOVA) tests were preformed on the electric fishing data; any significant results underwent post-hoc Fisher least significant difference (LSD) test. Residual and normal plots of the electric fishing data were examined and transformation of data was considered unnecessary. Non-parametric Kruskal-Wallis ANOVA tests were preformed on trap net data, which was not normally distributed (residuals examined). All analyses were conducted using Statistica versions 7 and 8 and Microsoft Excel.

3.3 Results

3.3.1 Climate in Hamilton City during study

Daily rainfall is displayed as a continuous line, while total monthly rainfall is displayed as points in Figure 3. High rainfall events are spread across the graph. There are large rainfall events in March, May, August and November 2007. The single largest rainfall event, within Hamilton City, occurred on 5 August 2007 when 41.8 mm of precipitation was recorded over a 24-hour period. High intensity rainfall events are likely to be more responsible for structural movement; but, sustained rainfall events may be more important for fish movement within streams. Sustained events occurred in July and August and are represented by high monthly rainfall results. Total monthly rainfall events are high in March and decrease to April, before increasing to a peak monthly rainfall in July.

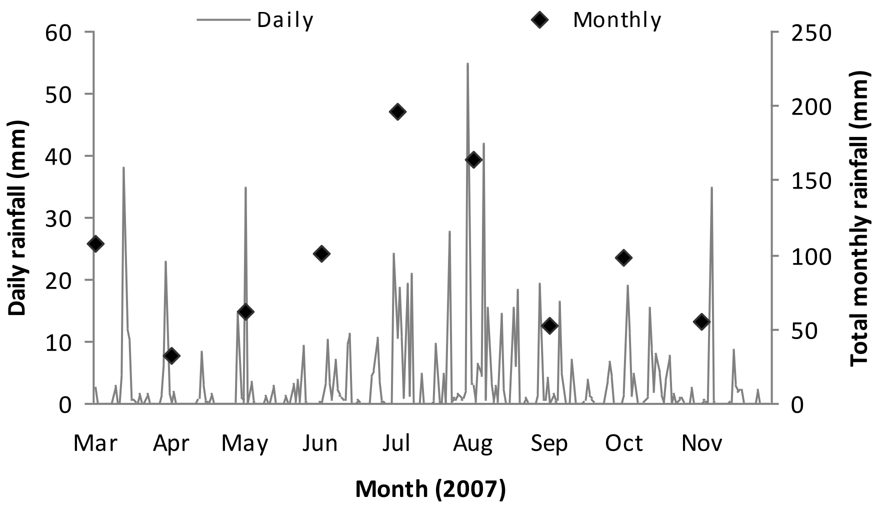

Figure 3: Daily and monthly rainfall (mm) for Hamilton City extracted from the http://cliflo.niwa.co.nz website. Data collected by the N.Z Meteorological Service (Hamilton International Airport) and extracted on 27 December 2007.

3.3.2 Structural movement and effectiveness

Data where visibility was poor and the efficiency of the structures could not be assessed were removed from the data-set (Figure 4). More logs were retained at sites than pipes and for a longer period of time, although both log and pipe effectiveness decreased over time. Over the eight months of monitoring, the greatest decrease occurred within the pipe section. Following rain events structures may have been obscured by an increase in sediment deposition rather than being washed from the reach. After the winter months the percentage of log structures retained increased slightly, and this may be the result of uncovering of structures previously covered by sediment.

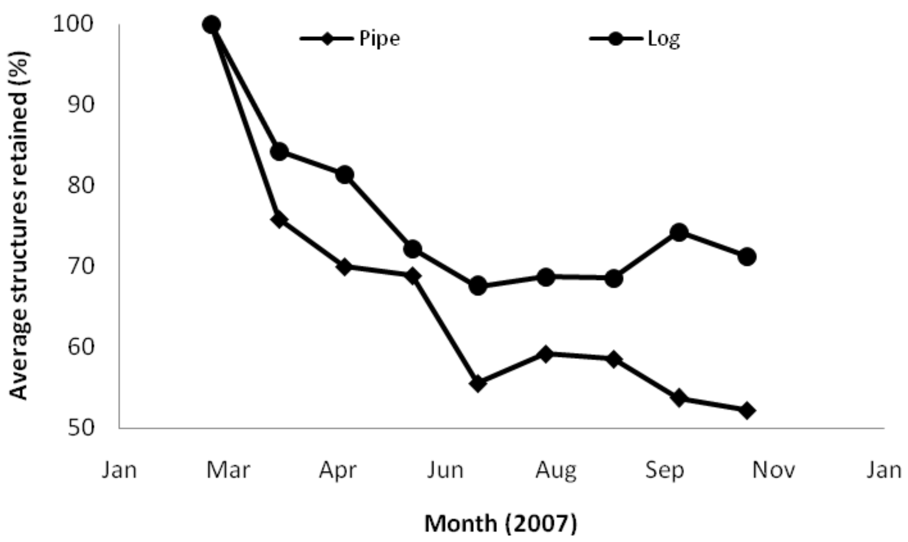

Figure 4: Average percentage of structures retained within each 20-m stream reach (n=10) from April to November. Confidence intervals are not shown for clarity.

Overall more structures were recorded as effectively providing habitat within the log section of the stream than the pipe section (Table 5). Effectiveness was assessed visually by the presence of a structure not completely covered with fine sediment. Initially 84% of logs and 75% of pipes were considered to be providing habitat. The greatest loss of log effectiveness, from start to end, was 30% at Site K10a. The greatest loss of effectiveness within the pipe section was 45% at Site M11. All logs retained their ability to provide habitat at sites K10 and P1. While all pipes, except at Site P1 where all pipes were retained, lost some of their ability to provide habitat. Logs maintained 71% effectiveness compared to 56% for pipes.

Table 5: Proportion of structures providing visible habitat at the first and last sampling periods (Jan-Nov 2007). Sites K10b and M12 have been omitted from table due to poor visibility.

Site	% logs effective		% pipes effective	
	Jan	Nov	Jan	Nov
K2	80	60	65	50
K10	90	90	80	75
K10a	80	50	80	45
K11	90	80	80	65
M1	90	70	75	60
M7	90	80	75	45
M11	70	60	80	35
P1	80	80	65	65
Average	84	71	75	56

3.3.3 Local organic matter and sediment changes

Coarse particulate organic matter (CPOM) refers to the coarse detritus, such as leaves and sticks, which accumulated on structures. The weighted % CPOM values are <13% in any month, therefore structures had a small effect on retention in these streams. More CPOM built up on logs than pipes, with markedly more initially found upstream of these structures (Figure 5). There was more CPOM upstream of log structures in May (autumn) than any other month sampled, presumably from leaf fall from deciduous trees upstream. The variability of CPOM upstream may have been caused by high flow events, for example, marked deposition upstream of logs in May, could have been associated with a high flow event close to the time of sampling (Figure 3).

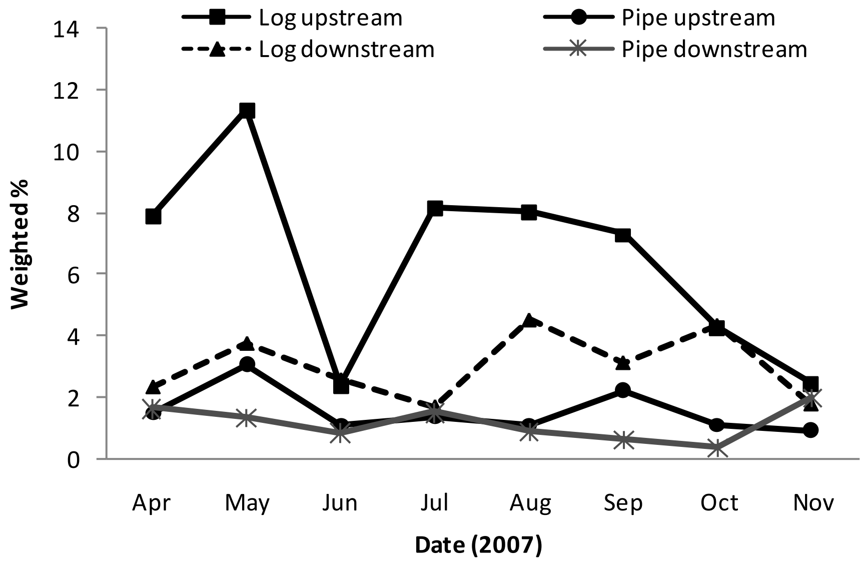

Figure 5: Average weighted percentage of coarse particulate organic matter build-up on log and pipe structures recorded upstream and downstream (April to November).

Scour refers to the removal (erosion) of fine sediment by flowing water and was recorded as the percent of sediment removed below the base of a structure (Figure 6). There was a gradual increase in scour from April to November over all structures, although weighted percentage values were low (<10%). Scour was more obvious upstream of the enhancement structures and, as with CPOM, was more pronounced on log structures than pipes, although differences were small.

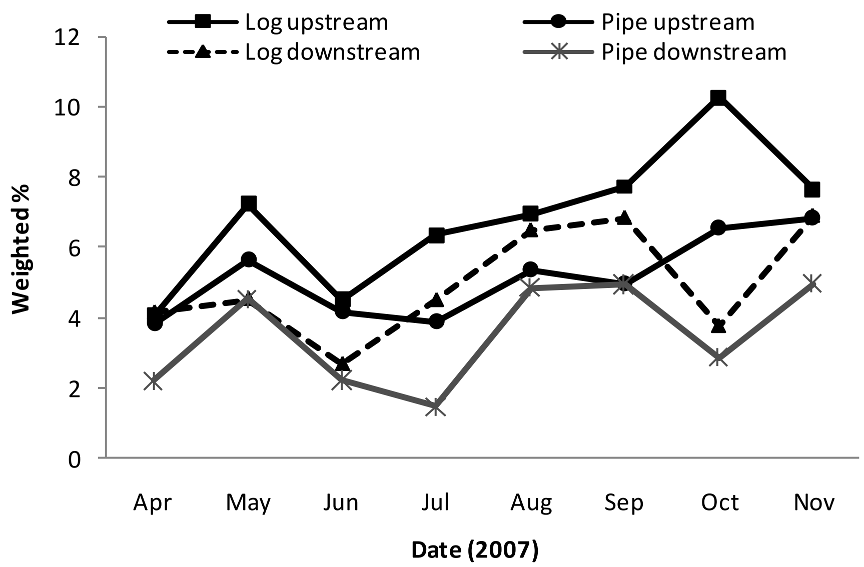

Figure 6: Average weighted percentage of scour recorded upstream and downstream of log and pipe structures (April to November).

The amount of fine sediment deposited on structures is presented in Figure 7. There was a gradual increase in deposition over time from April to November. Deposition was greater downstream of the structures rather than upstream, and was greatest downstream of the log structures, although observed differences were small. The area affected was greater than for scour (<25%) but may possibly reflect the redistribution of sediment from erosion upstream of logs, as well as sediment borne particles dropping out of the water column due to reduced current velocities downstream. There was a decrease in the amount of downstream deposition in September which may be due to sediment removal from higher than average flow rates, caused by winter flooding in August (Figure 3).

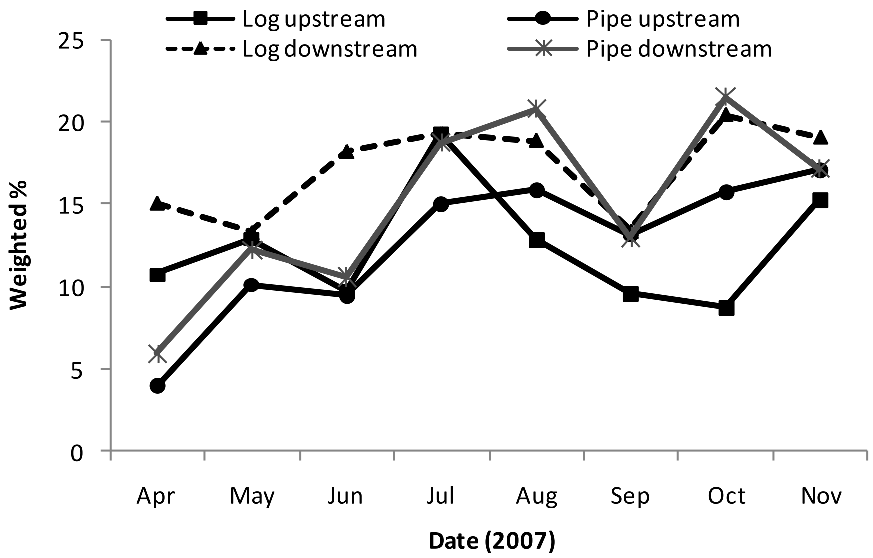

Figure 7: Average weighted percentage of fine sediment deposition upstream and downstream of log and pipe structures (April to November).

3.3.4 Fish populations within study sites

Site P1 had high species diversity, with banded kokopu, catfish, common bully, mosquitofish, giant kokopu, shortfin and longfin eel, and rainbow trout (one in summer 2006) present. Sites M1 and K10b also had a high number of species (5-7 species on any date). The biomass and density data in Table 6 are consistent across seasons for most of the streams sampled. However, there appears to be a general increase in winter with less fish caught over summer 2006 and 2007. In general, more fish were recorded during the summer 2006 surveys than the summer 2007 survey which could be due to time differences. The summer 2006 surveys were conducted from December 2006 to February 2007, while the summer 2007 survey was undertaken in November.

Density and biomass were greatest at sites K2, P1 and M7 where large numbers of shortfin eels were caught. A large number of shortfin eels were also caught at K10a but the size of the stream resulted in relatively low density and biomass results per unit area. Longfin and shortfin eel numbers remained constant over season. Higher numbers of longfin eel were caught at sites M1 and M7. Smelt numbers were high in the Kirikiriroa catchment in winter 2007; this probably indicates recruitment, as well as accessibility to stream sections, which do not have any major fish barriers downstream to the Waikato River. Smelt were not found at Site K11 in winter 2007 which has a perched culvert downstream under Glenn Lynn Avenue. Giant kokopu and longfin eel were present at several sites but occurred in low numbers (Table 6).

Whitebait species were caught in summer 2006 and summer 2007 indicating recruitment into urban streams at this time of year. Inanga have not been included in this table because of the low numbers caught.

Table 6: Number of wild fish caught per 60-m site by electric fishing (n=3) (a) summer 2006, (b) winter 2007, (c) summer 2007.

a. Summer 2006

Wild fish	K2	K10	K10a	K10b	K11	M1	M7	M11	M12	P1
Giant kokopu	0	0	2	1	0	0	0	0	0	2
Longfin eel	0	0	0	1	0	11	7	2	0	3
Shortfin eel	26	5	23	1	12	1	15	9	2	29
Smelt	0	0	1	15	0	0	0	0	0	0
No. of species	1	3	3	6	3	6	3	4	1	8
Total number	26	8	26	20	14	23	26	20	2	44
Density (no. 100 m^{-2})	132	48	50	105	56	70	89	39	3	119
Biomass (g m^{-2})	245	28	50	64	135	83	192	107	5	124

b. Winter 2007

Wild fish	K2	K10	K10a	K10b	K11	M1	M7	M11	M12	P1
Giant kokopu	0	0	0	1	3	0	0	0	0	4
Longfin eel	0	0	1	1	0	5	9	0	0	2
Shortfin eel	31	2	24	9	10	0	18	2	23	29
Smelt	2	2	26	32	0	4	0	0	0	0
No. of species	2	2	3	5	2	5	2	2	2	9
Total number	33	4	51	45	13	17	27	3	24	58
Density (no. 100 m^{-2})	161	32	102	218	64	62	92	34	34	177
Biomass (g m^{-2})	282	41	84	62	109	74	266	31	92	98

c. Summer 2007

Wild fish	K2	K10	K10a	K10b	K11	M1	M7	M11	M12	P1
Giant kokopu	1	0	0	1	0	1	1	0	3	1
Longfin eel	0	0	0	1	0	6	8	0	1	1
Shortfin eel	24	3	15	6	10	2	20	3	31	43
Smelt	0	0	0	5	0	0	0	0	0	0
No. of species	2	1	1	7	1	6	3	1	4	7
Total number	25	3	15	16	10	18	29	3	36	56
Density (no. 100 m^{-2})	124	17	29	76	48	67	100	34	39	159
Biomass (g m^{-2})	188	9	43	79	82	54	240	25	109	100

The density and biomass results for the trap net data (Table 7) were lower than those obtained by electric fishing (Table 6) for most sites. The exceptions to this were K10 where large numbers of inanga and smelt were caught resulting in high densities, and K10a, K10b and M12 which had higher biomass as a result of the use of fyke nets. Shortfin and longfin eels were caught in lower numbers with trap nets, whereas giant kokopu were recorded in higher numbers using trap nets, with 12 recorded (sampled with replacement) at Site K11. Mosquitofish were captured at Site K10 in summer 2006, but were absent in summer 2007. Pest fish captured were removed without replacement. Koura were recorded at three of the 10 sites sampled; these were K10b, M1 and M11. Several fish were caught in low numbers, notably rudd (one fish was captured at K2), torrentfish (captured at M1), and banded kokopu, catfish and common bully which were recorded at two to three sites.

Most fish species were caught in higher numbers at the initial fishing and in April and May (Table 7b). Whitebait were caught during the initial and November surveys in near equal numbers, and the November to February period appears to be when whitebait recruits enter Hamilton urban streams confirming results from electric fishing. The total number and density of fish was greatest at the initial fishing, and then decreased through the cooler months before increasing in November. Total biomass was low in May even though density was high, and this could reflect the high number of inanga and smelt caught, and the absence of eels which can provide substantial biomass.

Table 7: Number of wild fish species and koura caught per 10 urban stream sites using trap nets. Combined total for 9 trapping occasions between December 2006 and November 2007 using 15 Gee minnow traps at each site and fyke nets at K10a (3 nets), K10b (1 net), and M12 (3 nets). (a) Number of fish per site, (b) number of fish per fishing event.

(a) Fish per site

Wild fish	K2	K10	K10a	K10b	K11	M1	M7	M11	M12	P1
Giant kokopu	1	0	6	5	12	0	0	0	1	1
Inanga	11	51	4	4	0	2	0	0	0	0
Longfin eel	0	0	1	1	0	0	0	0	3	0
Shortfin eel	2	1	13	4	1	0	1	0	12	2
Smelt	5	49	0	10	0	1	0	0	1	0
No. of species	8	6	5	8	3	6	6	3	5	5
Total number	37	129	26	32	19	10	6	7	19	14
Density (no. 100 m^{-2})	174	799	54	146	82	35	21	3	30	39
Biomass (g m^{-2})	6	20	93	96	6	1	15	2	91	2

(b) Fish per sampling

Wild fish	Dec-Feb	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov
Giant kokopu	3	8	6	2	3	1	2	1	0
Inanga	21	25	14	2	4	4	1	0	1
Longfin eel	1	1	0	1	1	0	0	0	1
Shortfin eel	7	9	0	0	1	1	6	5	7
Smelt	2	25	25	6	3	4	0	0	1
No. of species	10	8	3	6	6	9	6	4	6
Total number	77	74	45	15	13	16	13	9	28
Density (no. $100m^{-2}$)	401	389	248	45	46	67	36	30	122
Biomass (g m^{-2})	89	66	7	13	23	8	33	43	48

3.3.5 Fish response to habitat

The electric fishing and trap net data for species per habitat suggests higher densities overall in the log and control sections than the pipe section (Table 8), although this is not statistically significant. Most of the species appeared to have relatively consistent numbers across treatments, including whitebait. The electric fishing data suggested a preference for the log section of stream by giant kokopu, but this was insignificant when fish numbers and habitat were compared (p=0.89). Trap net data suggested higher numbers of mosquitofish in the log section, but again this was statistically insignificant (p=0.13), and is may be due to the mosquitofish being predominantly from one site (K10). Koura appeared to be more common in the log and pipe sections of stream indicated by both fishing methods (Table 8), but this was insignificant for both electric fishing and trap net data (p=0.51 and p=0.57, respectively). Although biomass was twice as high in the control section of streams using trap nets, this was found to be insignificant using Kruskal-Wallis test.

Table 8: Number of wild fish and koura caught at 10 sites in each habitat treatment during (a) the three electric fishing surveys (summer 2006, winter 2007 and summer 2007), and (b) trap nets sampled monthly from April to November. Units are per 20-m reach except for density and biomass which are standardised to 100 m^{-2} and g m^{-2}, respectively. Control section across all streams included three fyke nets, while log and pipe sections had two fyke nets.

(a) Electric fishing results

Species	Log	Pipe	Control
Giant kokopu	21	9	12
Longfin eel	23	15	21
Koura	10	8	4
Shortfin eel	170	115	146
Smelt	31	31	25
No. of species	11	9	11
Total number	281	200	236
Density (no. 100 m^{-2})	872	620	814
Biomass (g m^{-2})	1141	841	1109

(b) Trap net results

Species	Log	Pipe	Control
Giant kokopu	25	16	26
Inanga	23	10	41
Longfin eel	11	6	12
Koura	8	9	4
Shortfin eel	67	64	62
Smelt	20	32	30
No. of species	11	10	10
Total number	200	165	201
Density (no. 100 m^{-2})	514	411	458
Biomass (g m^{-2})	65	73	194

Fish density and biomass in summer 2006 and summer 2007 are near equal for the log and control sections (Figure 8). There was no statistically significant result for habitat compared to either biomass or density for electric fishing using one-way ANOVA.

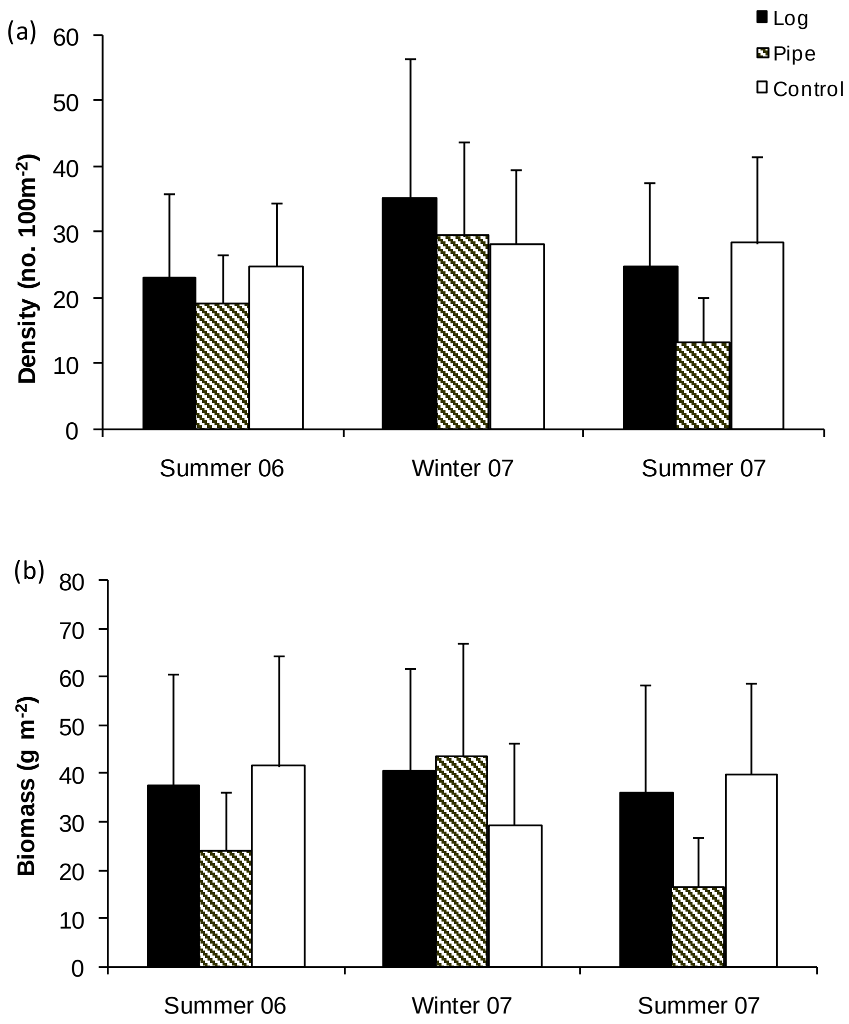

Figure 8: (a) Density and (b) biomass (means ± 95% confidence interval) of all wild fish caught by electric fishing within each habitat treatment (n=10).

Biomass per treatment at each of the separate sites was highly variable (Figure 9). The elevated biomass at Site M11 during the summer 2006 survey was insignificant ($p=0.15$) and was probably the result of one large longfin eel (measuring 850 mm and 1820 g), which was relocated within the catchment before juvenile giant kokopu release. This eel was removed because eels of this size are known to be piscivorous on smaller fish. Site M12 shows a significant increase in biomass from summer 2006 to winter 2007 ($p<0.01$). Logs appeared to provide habitat for fish species in winter at sites K2, K10 and K10b. The higher biomass is likely to be a result of smelt recruitment into the streams. Pipes appeared to provide habitat for fish in winter at sites K10a and M7 this seemed to be due to smelt and shortfin eel movement into the pipe section.

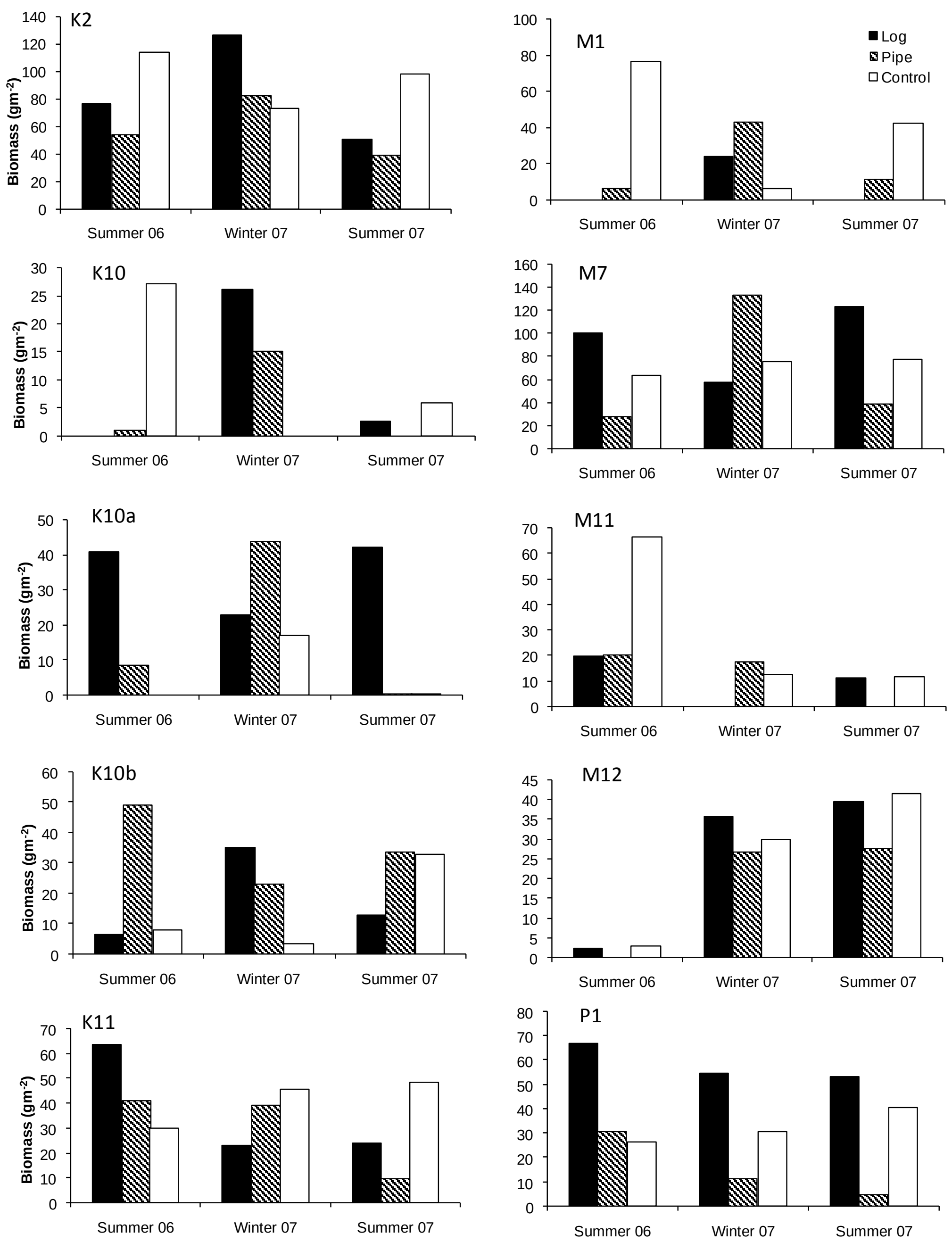

Figure 9: Total biomass of wild fish per site (n=10) at each treatment. Fish were captured by electric fishing in summer 2006 (prior to restoration), winter 2007 and summer 2007 (note y axis varies).

Density and biomass data for fish caught using trap nets was varibale across treatments and monitoring events (Figure 10), with the difference being statistically significant between biomass and seasons ($p<0.05$) using Kruskal-Wallis (including fyke nets). Biomass and density data were markedly lower in the cooler winter months. The error bars were slghtly higher for the biomass data which may be the result of a low number of large fish caught at some site (e.g. large eels). Fish density and biomass were lower than for the electric fishing surveys. The elevated density and biomass in the control section in June, July and August is probably caused by consistent captures of inanga and/or smelt at Site K10.

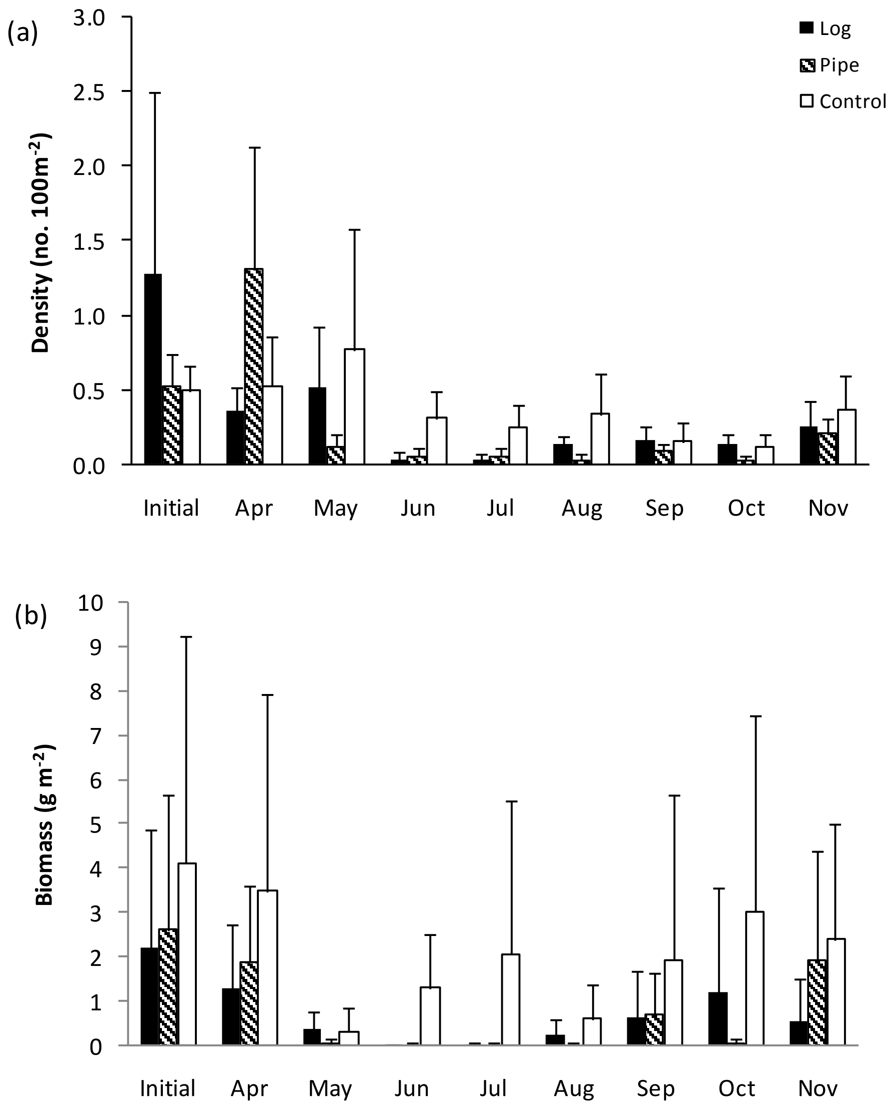

Figure 10: (a) Density and (b) biomass of all wild fish (means ± 95% confidence interval) caught using trap nets within each 20-m habitat treatment (n=10) (initial = November 2007 to February 2008).

3.4 Discussion

Log structures used to enhance habitat in Hamilton urban streams were considered to be more effective than pipes at providing habitat, due to fish reaction, retention and microhabitat alterations. Overall fish biomass and densities were greater in the log section of stream than the pipe section. A greater percentage of logs were retained for longer periods of time, and although differences were small, CPOM build-up, scour and deposition were greater around the log structures than pipes.

CPOM was greater upstream of log structures, especially in autumn when deciduous trees shed their leaves, and is important because it is known to support macroinvertebrate communities that can provide a food source for fish (Richardson and Neill, 1991). CPOM can influence other stream processes (Wallace et al., 1997), including enhancing microhabitat by fine sediment retention and substrate stabilisation (Scarsbrook and Townsend, 1994). Macroinvertebrate productivity was improved through leaf litter retention in habitat artificially enhanced using boulder clusters by Negishi and Richardson (2003).

Scour improves stream bottom variability contributing to microhabitat formation, and was more pronounced upstream of the log structures shortly after installation and again during high winter flows. Scour around logs is important for providing fish habitat, especially at critical times of their life cycle (Borg et al., 2007). Deposition was found to be greater downstream of structures where the velocity was slowest, and increased over time contributing to infilling and habitat loss. While more deposition was recorded downstream of log structures, the hollow part of the pipes was observed to infill and pipes were more likely to be completely covered in sediment reducing their ability to provide habitat. Large woody debris influenced sediment storage and transport in headwater streams in south-eastern Alaska (Gomi et al., 2001). The highly dynamic nature of urban stream hydrology (Paul and Meyer, 2001) suggests that structures may not be a permanent habitat feature of the restored streams.

The rainfall data collected for Hamilton City from the Hamilton International Airport, south of the city centre, suggested that structural movement was more related to high daily rainfall levels than sustained rainfall events. Impervious cover of urban catchments results in increased surface runoff to streams, increasing flow volumes and velocities which then have enough power to move in-stream structures (Paul and Meyer, 2001). Higher velocities make stream channels less stable, and increase erosion and sediment movement, thereby contributing to the loss of in-stream cover and habitat (Wang et al., 2001). The high discharge of water within the Hamilton streams studied suggested that rainfall events would have aided in the apparent movement of structures, although some were also buried by sediment for variable periods. Pipe structures appeared to move the most during the study which may be partly due to the method used to secure structures. The one piece of reinforcing bar used to secure pipes to the stream bank may have been more easily moved than the two pieces of reinforcing bar used to secure logs to the stream bed. As well as structural movement, storm flows can have a detrimental effect on fish by increasing turbidity (Richardson et al., 2001), and displacing fish (Morgan and Cushman, 2005). Storm flows are likely to have a greater effect on juvenile fish and fish residing in scour prone habitats in urban environments (Roy et al., 2005). While single rainfall events appear to have a greater impact on structural movement, persistent rainfall may be more important for fish movement in urban streams.

The ten restoration sites had a range of fish species present with native fish dominating the fish community, and shortfin eel being the most abundant species (Wilding, 1998; Aldridge and Hicks, 2006). Smelt and inanga were captured in higher numbers in late summer to early autumn, while whitebait were caught from November to February indicating that they begin to enter streams, after migrating up the Waikato River in November. The density of fish caught in these urban streams was more variable than reported by Hicks (2003) by electric fishing three pasture streams of a similar size to streams in this study. In Hicks (2003) study densities ranged from 39 to 117 fish 100 m^{-2} and biomass ranged from 14 to 74 g m^{-2}, compared to densities in this study of 3 to 177 fish 100 m^{-2} and biomass of 5 to 282 g m^{-2} in the present study. Biomass was highest in both studies where a large number of eels were caught. These results indicate that density is similar but more variable than found in pasture streams, but biomass can be much higher in urban streams.

The results from the single-pass electric fishing data appeared to be consistent across seasons while trap net capture rates decreased over winter. Fish are known to be less active at lower temperatures (David and Closs, 2003) and sampling method probably accounts for the seasonal differences. Trap nets bias the size of the fish that can be captured, such that only small fish can be captured in Gee minnow nets while larger fish are captured in coarse-mesh fyke nets. The use of two types of trap nets was necessitated by the variable sizes of the streams. Use of fyke nets at only a limited number of sites (where stream depth allowed) resulted in higher biomass at those sites, where larger fish could be caught. Predation was observed in trap nets (Gee minnow and fyke nets) where piscivorous fish (such as eels) were caught with smaller fish or crayfish (Aldridge and Hicks, 2006). These authors recorded a relatively high proportion of zero catches in Gee minnow nets, which was also found during this survey. Although electric fishing is labour intensive and can adversely affect fish and habitat (McCullough and Hicks, 2002), most sizes of fish can be caught and the effect of predation is avoided. Due to the frequency of sampling required and the risk of disturbing fish and habitat treatments, electric fishing was not used as the sole method of survey, and trap nets were used to supplement data. The use of a combination of methods provided additional information about fish populations that a single method would have missed. Disruptive as it can be, electric fishing overcame the limitations of low water temperature associated with trapping, and appeared to provide the best synoptic view of fish abundance.

Habitat restoration has been used successful for many fish species in different countries. The main habitat structures used for this purpose have been large wood and boulder clusters (De Jong et al., 1997; Bonnett et al., 2002; Negishi and Richardson, 2003; Mutz, 2004). In this study most fish species appeared to slightly prefer the log section of habitat over the pipe section, although this result was statistically insignificant, reflecting variability probably brought about by interactions with stream characteristics such as size and turbidity. An apparent preference for log structures was seen in the density and biomass of fish, and the number of giant kokopu (over treatments), but these were also insignificant using ANOVA. Giant kokopu are known to prefer debris dams and cover (Bonnett and Sykes, 2002; Baker and Smith, 2007), which the logs may have provided. Koura are known to associate with cover in streams, including woody debris, undercut banks, leaf litter and tree roots (Jowett et al., 2008); in the present study koura were in higher abundances in the log and pipe sections. Although few results were statistically significant, habitat restoration using logs appeared to be partially successful at the scale carried out in this study, providing at least short-term benefit to fish populations within small Hamilton urban streams.

Chapter 4 Survival and retention of released farm-reared Giant kokopu

4.1 Introduction

The native giant kokopu (*Galaxias argenteus*) is considered threatened with populations in gradual decline (Tisdall, 1994). This decline is probably a result of habitat reduction due to forest clearance, wetland drainage and urban development. Barriers to migration also threaten inland diadromous populations by restricting range, although diadromy is not obligatory within this species (David et al., 2004). Habitat preference of giant kokopu has been extensively studied for conservation management purposes. The developed nature of general habitats occupied by the fish (i.e., swamps, lowland coastal areas and wetlands) has resulted in the study of more specific microhabitat selection. A study of the giant kokopu records on NIWA's Freshwater Fish Database (FFDB) by Bonnett and Sykes (2002) indicated that five habitat features were important for giant kokopu: in-stream cover, deep water, low water velocity, proximity to the sea, and overhead shade/riparian cover. In-stream cover in the form of debris dams and undercut banks in pools and backwaters was also found to be important for habitat of giant kokopu in first and second order streams draining the Hakarimata Range in the Waikato Region (Baker and Smith, 2007). Whitehead et al. (2002) noted that the preferred microhabitat position was likely to be in low velocity pools (to reduce energy expenditure) and beside a fast-flowing current or riffle to capture drifting invertebrate food sources. Although general habitat selection of juvenile and adult fish appears to be similar (Bonnett and Sykes, 2002), these preferences are different at the microhabitat scale, where small fish occur in shallow backwaters near faster flows, whilst larger fish prefer deeper, slower flowing pools (Whitehead et al., 2002). This segregation in habitat may be due to conspecific competition where the heaviest individuals hold the most desirable position within a pool (David and Stoffels, 2003).

Odours have been used to determine conspecific relationships for several of the whitebait species (*Galaxias* spp.). Baker and Hicks (2003) found that juvenile migrating banded kokopu, and to a lesser degree inanga and koaro, were attracted to other galaxiid species and conspecific odours. Baker and Montgomery (2001) have shown that juvenile migratory banded kokopu are attracted to adult odour at intermediate concentrations, but attraction was retarded at high concentrations. The authors suggested that this attraction may be used (by migrating juveniles) as the basis for selecting suitable habitat and population densities. Giant kokopu are nocturnal, and therefore probably rely on odour as a means of species interaction. A study based on juvenile inanga (*Galaxias maculatus*) found that these fish avoided the odour of shortfin eels in tank trials (McLean et al., 2007).

Electric fishing (Baker and Smith, 2007) and spotlighting (David and Closs, 2002; Whitehead et al., 2002; David and Closs, 2003) have been used to monitor populations of giant kokopu. The method used to monitor populations depends on the information required. Electric fishing is useful where a single fishing survey is required and to indicate daytime habitat of fish (Baker and Smith, 2007). However, electric fishing can have a detrimental effect on fish and habitat, and is labour intensive (Mesa and Schreck, 1989). Spotlighting is a useful method of observing behaviour, movement, and active night time habitat selection of nocturnal giant kokopu (David and Closs, 2002; Whitehead et al., 2002; David and Closs, 2003). However, spotlighting is only able to be conducted under suitable water quality (clear) and depth (stream bottom visible) conditions. Trap nets are not a commonly used method for assessing giant kokopu populations, but adult fish have been recorded in fyke nets (Jellyman, 1979) and juveniles in Gee minnow nets (Aldridge and Hicks, 2006).

Giant kokopu are a natural component of fish populations in Hamilton urban streams where they were captured at five of 41 sites surveyed during the 2005/06 summer (Aldridge and Hicks, 2006). Physical characteristics of Hamilton urban streams where giant kokopu were caught varied widely from 0.75 to 1.75 m in width, 0.15 to 0.80 m in depth, and clear to highly turbid waters. All sites had some degree of shading (Aldridge and Hicks, 2006). Giant kokopu were chosen for stocking into urban streams because (i) they were already present in urban streams at low densities, (ii) previous work suggested habitat preferred by this fish could be enhanced by introducing physical structure to streams, (iii) they have a threatened conservation status, and (iv) farm-reared fish could be easily obtained. This chapter discusses the reaction of these farmed fish to both introduction and habitat enhancement in five urban streams.

Aldridge and Hicks (2006) found giant kokopu in association with longfin eels, shortfin eels, banded kokopu and inanga within Hamilton urban streams. Giant kokopu are a lowland fish (McDowall 2000), and as such are often found with other lowland species. The association of giant kokopu with the shortfin eel is of interest because eels are a native piscivore, and were the dominant species found (per site) in Hamilton urban streams (Aldridge and Hicks, 2006).

Giant kokopu are known to occupy a restricted home range (David and Closs, 2003), and it was predicted that juveniles introduced to the stream would find suitable habitat to occupy and then stay within a home range. The home range of adult giant kokopu in small South Island streams in New Zealand rarely exceeded 26 m (David and Closs, 2003). An enhanced habitat reach of 20-m was therefore considered to be long enough to retain and provide a habitat response in juveniles.

The aims of the work presented in this chapter are to:

1. Determine the retention of juvenile fish in the study reaches,
2. Measure growth rates of the introduced population,
3. Assess any preferences of juvenile giant kokopu in relation to habitat enhancement (see Chapter 3), and
4. Explore the relationship between farm-reared giant kokopu and wild fish populations, particularly wild giant kokopu and piscivorous eels. In addition a comparison of sampling methods for catching juvenile giant kokopu is made at one of the sites.

4.2 Methods

4.2.1 Procurement of fish

Juvenile giant kokopu were obtained from the whitebait farm of Charles Mitchell and Associates Ltd. in Te Uku, Raglan, New Zealand. The juvenile fish had been spawned and reared in captivity from a brood stock of 25 individuals (therefore not sea run), and as a result had limited genetic variability. The juvenile giant kokopu were reared in outdoor ponds until November 2006 after which they were captured and transferred to 200-L plastic drums fitted with airlift powered sponge filters. Initial food consisted of frozen calanoid copepods, with gradual weaning onto increasing amounts of artificial salmon starter crumbles. All fish were held in tanks with fish of the same species and of a similar size. Fish were considered to be naïve, having never experienced predation, hydrological variability, or habitat and food selection pressure.

4.2.2 Holding of fish in the laboratory

Fish were held at The University of Waikato fish laboratory prior to release. In these new surroundings fish initially experienced high mortality after several days. Fish were visually inspected for signs of disease and were recorded as suffering from white spot, fin root, and gill diseases. Fish were treated with the appropriate chemicals and doses, but mortality continued. Many of the diseases suffered were not present in their original surroundings, but were present in the University fish lab, which housed a multitude of different fish species, many of which were introduced. Stress from transportation and high tank water temperatures (often above 20°C) in February and March may have increased their susceptibility to illness. To decrease the mortality rate, new fish were marked and held at the whitebait farm, before being released directly to streams, this eliminated death by disease, prior to release.

4.2.3 Marking fish

Stocked fish were marked in sterile laboratory conditions by injecting a fluorescent elastomer acrylic mark beneath the surface layer of skin, above the anal fin (Figure 11). Each batch of 30 individual giant kokopu were removed from the holding tank and placed into a well-aerated bucket of water. The ingredients of the elastomer mark (colour and curing agents (Northwest Marine Technology Inc.)) were mixed together and stored in a chilled syringe ready for injection. Prior to marking, all fish were anaesthetised with laboratory grade benzocaine (ethyl aminobenzoate). Marking was carried out with a 0.3 cc injecting syringe, which was sterilised between each injection with 95% ethanol and saline solution, and replaced frequently. Each fish was weighed (± 0.1 g) and measured (± 1 mm) before being placed into a recovery bucket of well-aerated clean water. Fish were left to recover and assume pre-marking routines for at least three days before being released into streams. There was no recorded mortality as a result of marking the fish and marking had 100% retention prior to release. Elastomer marks were chosen because they could be used on small fish (<50 mm), were easy to administer, and fluoresce under ultraviolet (UV) light (Figure 11). Stocked fish caught in the field could be visually differentiated from the wild population with the aid of a hand-held UV torch if necessary.

Figure 11: Elastomer mark located above the anal fin of a juvenile giant kokopu (arrow). The bottom photo shows the elastomer mark under UV illumination.

4.2.4 Release of fish

At the time of release the fish were approximately six months old, with an average fork length of 60 mm (42 to 78 mm) and an average weight of 1.93 g (0.71 to 4.66 g). Giant kokopu were released at five of the ten restoration sites to determine survival, growth and habitat preference within Hamilton urban streams. Fish were released between 16 March and 3 April 2007; K11, K10a and K10b (16/03/07), P1 (19/03/07) and M11 (03/04/07), after a holding period of 3 – 21 days. Sites were randomly selected to avoid any bias created by the existing physical characteristics of the stream. In accordance with the Ministry of Fisheries Transfer number AK108a, fish were treated for parasites and disease with the recommended dose of formalin and either malachite green or methylene blue 24 hours prior to release (Appendix 1). The juvenile giant kokopu were transported in a lidded plastic bucket, three-quarters filled with their tank water, which was continually aerated. On arrival at the stream, the fish were placed in a lidded holding container with holes at either end covered in fine mesh gauze to promote water exchange. The holding container was placed into the flow of the stream so that water could slowly exchange and the fish were left to acclimatise to the new stream water and flow conditions for 5 hours (± 1 hour). After the acclimatisation period, the fish were released into the stream in groups of ten individuals, by tilting the bucket so that fish could swim out into the middle of each of the three stream habitat treatments (ponga log, clay pipe, and control; see Chapter 3). In total, 150 fish were released into streams.

4.2.5 Fishing methods and measurement

Treatment sites were extensively fished using trap nets (Gee minnow and fyke nets), and electric fishing prior to the release of farm-reared giant kokopu to determine existing populations (Section 3.2.3). Populations of stocked giant kokopu were monitored monthly using trap nets and spotlighting, and by electric fishing three months after release and at the termination of the study in November 2007 (as detailed in Section 3.2.3). Spotlighting was conducted between dusk and 5 hours after dark at three of the five stocked streams (K11, M11 and P1) which had sufficiently high water clarity/visibility for spotlighting to be successful. The spotlight was powered by a single 12-volt battery. Juvenile kokopu within the size range of the released giant kokopu were captured where possible for identification and measurement. Total length of recaptured fish (± 1 mm) was measured using a fish measuring board.

4.2.6 Statistical analysis

The highly variable catch rate has meant that a limited amount of statistical analysis could be carried out on the data. Densities and biomass of the fish have been displayed as means ± 95% confidence intervals (CI). Exponential regression analysis of the length and weight increase over time was used to determine growth rates of the released giant kokopu. A length-weight equation was calculated for juvenile giant kokopu, which are poorly discriminated in existing equations. The electric fishing results of the released giant kokopu distribution were tested using one-way analysis of variance (ANOVA), with any significant results undergoing a post-hoc Fisher least significant difference (LSD) test. Residuals and normal plots of the data were examined and were not considered to need transformation. The residuals and normal plots for the trap net and spotlighting data were examined and did not follow a normal distribution, therefore trap net data was analysed using non-parametric Kruskal-Wallis ANOVA by ranks. Spearman rank-order correlations were undertaken to test any relationship between spotlighting and trap net data on the same dates, and relationships between released giant kokopu and shortfin eels. All analyses were conducted using Statistica versions 7 and 8, and Microsoft Excel.

4.3 Results

4.3.1 Retention of released fish

Released fish were recaptured at four of the five release sites. Released fish were never recaptured at Site K10a, which was wide (2.49 m), deep (0.35 m), had a high density of shortfin eels and a high turbidity/suspended sediment load. One fish was recaptured three times (26/06/07, 27/07/07, 22/08/07) within the log and pipe sections of K10b using trap nets. The depth of the stream at Site K10b (0.30 m) may have limited the recapture rate of the released giant kokopu.

Retention of giant kokopu was greatest at the other three release sites (K11, M11 and P1) where monthly spotlighting observations were also coincidentally made. These sites had suitable water quality for spotlighting (i.e., the bottom of the stream could be seen due to a shallow water depth and clarity) and were relatively narrow. Released giant kokopu densities were initially highest in the log section of K11 but no fish were sighted after July (Figure 12). The disappearance of released giant kokopu after July at Site K11 may have been caused by high flows as a result of intense localised rainfall. Fish at P1 appeared to prefer the log section of stream ($p=0.01$) throughout the duration of the study, although there were no sightings from July to October (Figure 12). P1 is shallow (0.21 m), wide (2.13 m) and clear. The best retention of released giant kokopu was observed at M11, where significantly more giant kokopu were observed ($p=0.002$ and $n=90$), and fish where observed on every sampling occasion throughout the study. Site M11 is shallow (0.18 m), narrow (1.10 m), has a low density of shortfin eels, and clear water. There appeared to be a decrease in the density of fish at Site M11 in July and again in November, although this was statistically insignificant. It is likely that as fish grow the stream has a reduced capacity to support higher densities of fish, potentially leading to decline over time. Densities at Site M11 were highest in the control section of stream on most occasions which is probably due to existing habitat availability such as pools and undercut banks. This preference diminished over time and was almost equal in all habitat reaches by November.

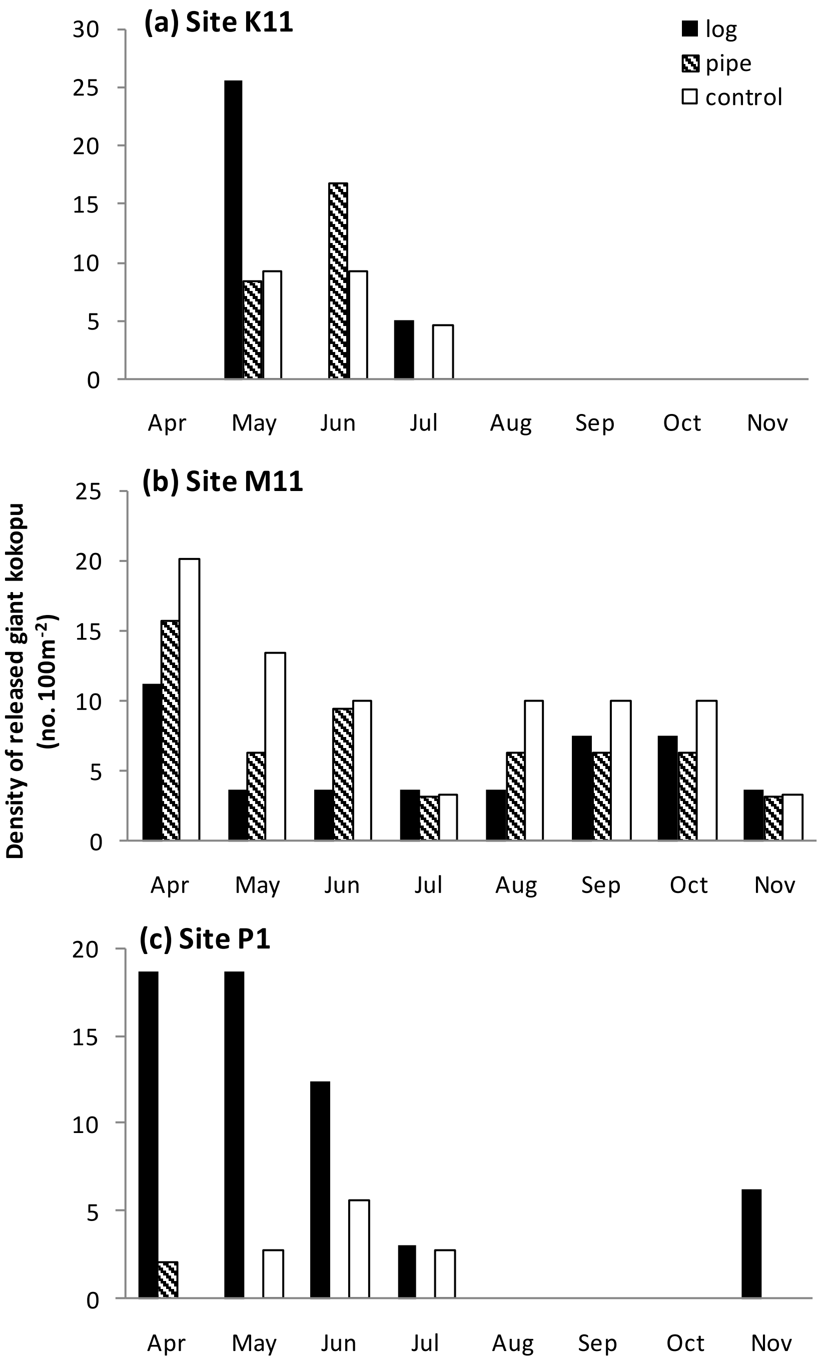

Figure 12: Density of released giant kokopu observed in each 20-m reach (from April to November 2007) by spotlighting sites K11, M11 and P1 (Note: y axis varies between sites).

4.3.2 Comparison of fishing methods

All three fishing methods were conducted at Site M11 which had a high retention of released giant kokopu, allowing a comparison of methods. Spotlighting data for this site are displayed in Figure 12 (only densities are displayed as fish were often unable to be caught and measured). Electric fishing data are limited by the number of fishing surveys undertaken. The intensive effort required, risk of habitat disturbance, and potential detrimental effect of electric fishing on juvenile fish limited the number of electric fishing surveys to three times throughout the duration of the research, including two surveys after giant kokopu release.

Trap net biomass increased sharply over time especially in the control reach of the stream (Figure 13). Spotlighting indicated a decrease in density in July, but trap nets did not show the same decline. The number of stocked giant kokopu caught using trap nets and spotlighting on the same occasions were compared using an Spearman rank correlation, which indicated a slightly negative relationship (r_s= -0.38). Higher densities in the trap net data for August were not observed in spotlighting results (Figure 12). Spotlighting densities were higher at the beginning of the study and decreased over time while trap net densities do not show this pattern.

The electric fishing data shows a similar result to the spotlighting data when comparing giant kokopu and shortfin eels (refer to Figure 17). There were less giant kokopu captured from the electric fishing data suggesting that spotlighting is a better method for observing giant kokopu if water clarity permits. Considerably more shortfin eels were recorded by electric fishing, which shows that this is a better method for capturing eels.

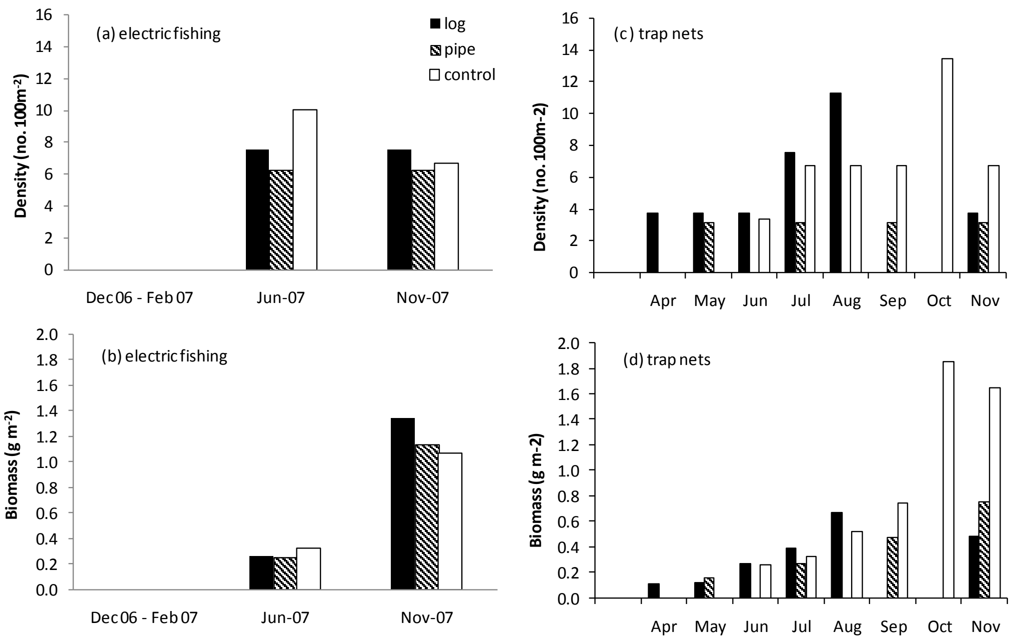

Figure 13: Biomass and density data for stocked giant kokopu at Site M11 captured by electric fishing and trap nets (five Gee minnow nets per 20-m habitat reach). Spotlighting data are not presented here because not all fish seen were caught for weighing (see Figure 12).

4.3.3 Growth of released giant kokopu

The length and weight of the juvenile giant kokopu measured have been displayed as a regression in Figure 14 (n=283). These data were used to calculate a length-weight relationship for juvenile giant kokopu measuring from 42 to 78 mm, i.e.,

$$W = e^{-12.60} L^{3.232},$$

Where W = weight in g and L = length in mm.

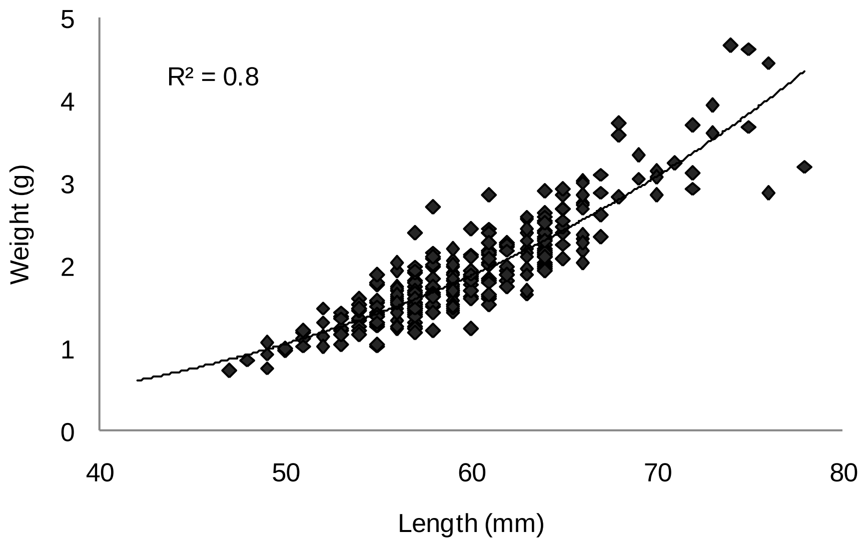

Figure 14: Length-weight relationship of farm-reared juvenile giant kokopu measured prior to marking (n=283).

Released giant kokopu increased in length and weight at all sites where they were caught, over an eight month period (Figure 15). A one-way ANOVA and post-hoc LSD revealed that the length of the fish at P1 was significantly different to the other sites at release ($p<0.05$), but fish were held in the lab for several weeks prior to release where growth was possible. The growth data includes records from all sampling methods and habitats where fish were caught and measured. Between one and seven individuals were caught per sampling survey, with seven fish captured twice at M11 in October (22/10/07 and 30/10/07). Individual growth response could not always be determined because the fish were batch rather than individually marked. However, at Site M11 individual fish appeared to be found in the same approximate locality (i.e., within the same pool) suggesting individual site fidelity.

The length and weight data of introduced fish increased substantially from March to November. Mean growth sometimes decreased on consecutive samplings due to different combinations of individuals being caught. Stocked fish caught in November at P1 had experienced substantial growth, with the larger of the two individuals caught measuring 132 mm in length. The average growth rate per day varied over the four sites; growth rate in length was lowest at Site K10b (0.19 mm day^{-1}) and greatest at Site K11 (0.33 mm day^{-1}) (Table 9). However, Site K11 had limited data, because fish were not recaptured after July, which would influence growth rate estimates because fish length increases faster at a smaller size. Average weight gain was lowest at Site K10b (0.03 g day^{-1}) and greatest at Site P1 (0.11 g day^{-1}), where weight was calculated from the length-weight regression for giant kokopu in Table 4 (Chapter 3).

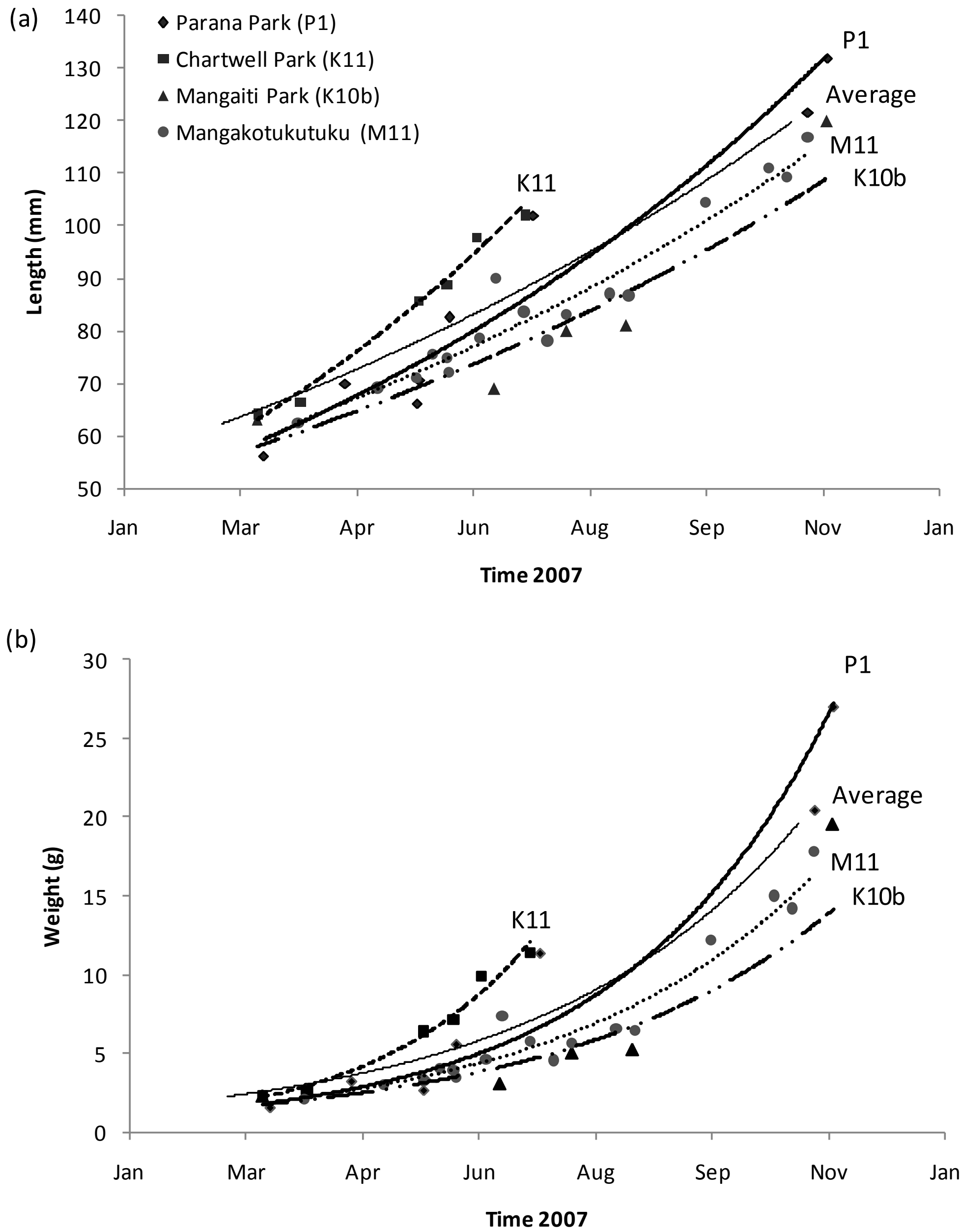

Figure 15: Growth of released giant kokopu at individual stream sites and overall sites (Average). Site exponential regression lines are presented for (a) average length and (b) average weight. Data were collected from marked fish using all fishing methods combined.

Table 9: Average daily length and weight increase of juvenile giant kokopu at each site (calculated from the initial to the final length) and weight changes, divided by the number of day's fish were retained in streams (-no fish recaptured).

Site	Length increase (mm day^{-1})	Weight increase (g day^{-1})
K11	0.33	0.08
K10a	-	-
K10b	0.19	0.03
M11	0.22	0.06
P1	0.31	0.11
Average	0.31	0.07

4.3.4 Habitat use

Habitat use by released giant kokopu was determined using single pass electric fishing over the five release sites. Initial density and biomass were calculated from the 10 individual fish released into each treatment reach per site in March and April; values vary between treatments as the area of each reach differed, due to variations in width. From June to November density decreased while biomass increased (Table 10). Despite similar densities across treatments initially, there was a slightly higher density of giant kokopu within the log section (than the other two sections) in June and this was maintained in the November sampling survey. The biomass in June was similar across the three 20-m restored habitat treatments (log, pipe, control). Biomass then increased in November suggesting that the stocked giant kokopu grew at a faster rate within the log sections of the stream, but there is also a slightly higher density which effects biomass. The log section has almost double the biomass of the control and pipe sections in November (Table 10). However, due to the large amount of variability in the data this result was insignificant (p=0.25).

Table 10: Average total (a) density (no. 100 m^{-2}), and (b) biomass (g m^{-2}) of released giant kokopu measured by single-pass electric fishing (n=5). Initial March-April 2007 data were determined from the release of 10 juvenile giant kokopu into each habitat treatment (i.e. 30 fish per stream section) and represent initial density or biomass. Values in parentheses are based on the proportion of the initial values.

(a) Density (no. 100m^{-2})

Habitat	Mar-Apr	Jun	Nov
Log	36.75 (100%)	5.01 (14%)	2.75 (7%)
Pipe	32.36 (100%)	1.26 (4%)	1.26 (4%)
Control	35.79 (100%)	2.57 (7%)	1.34 (4%)

(b) Biomass (g m^{-2})

Habitat	Mar-Apr	Jun	Nov
Log	0.95 (100%)	0.26 (27%)	0.53 (56%)
Pipe	0.84 (100%)	0.17 (20%)	0.23 (27%)
Control	0.92 (100%)	0.16 (17%)	0.21 (23%)

4.3.5 Relationship with wild fish populations

4.3.5.1 Relationship with shortfin eel

The most numerous species by density and biomass within Hamilton urban streams was the shortfin eel. The common occurrence of this potentially piscivorous fish meant that it was likely to have an interaction with the released juvenile giant kokopu. Within the control section of Site M11, a marked giant kokopu that had been attacked was captured in a Gee minnow net collected on the 26/10/07 (Figure 16). It is reasonable to suggest that the fish was attacked prior to capture as no other fish were caught in the same trap net. A shortfin eel was observed within the same reach of the stream four days later (30/10/07) while spotlighting, and was likely to be responsible for the attack.

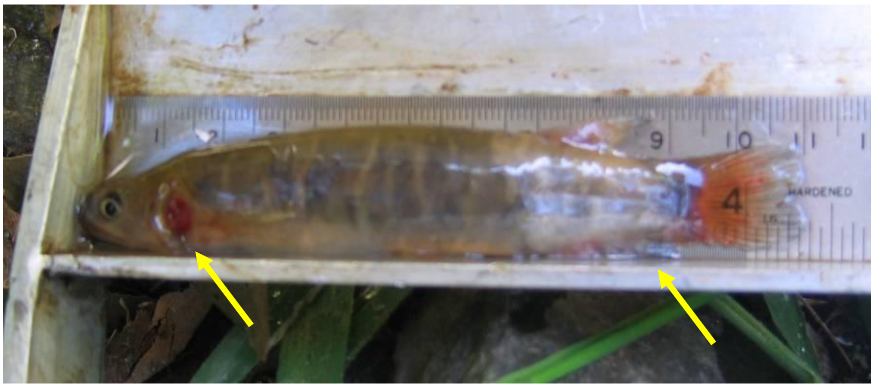

Figure 16: Attack on a marked giant kokopu caught in a Gee minnow net at Site M11. Arrows show areas of injury.

The relationship between eel and juvenile giant kokopu was examined at three of the five release sites (K11, M11 and P1). Densities of released giant kokopu and shortfin eel were calculated per 20-m restoration reach for these three sites (Figure 17). There was a weak negative correlation between shortfin eels and giant kokopu r_s = -0.13 over sites K11, M11 and P1, but no clear relationship across treatments. Highest densities of giant kokopu were recorded at M11 where two large eels (>700 mm) were removed from the study reach. The log section at P1 seemed able to support higher densities of both species which may indicate that the complex habitat provided by the logs may have allowed the eels and giant kokopu to co-exist with reduced competition. Additionally there may have been more food available as a result of the organic matter surrounding the log structures.

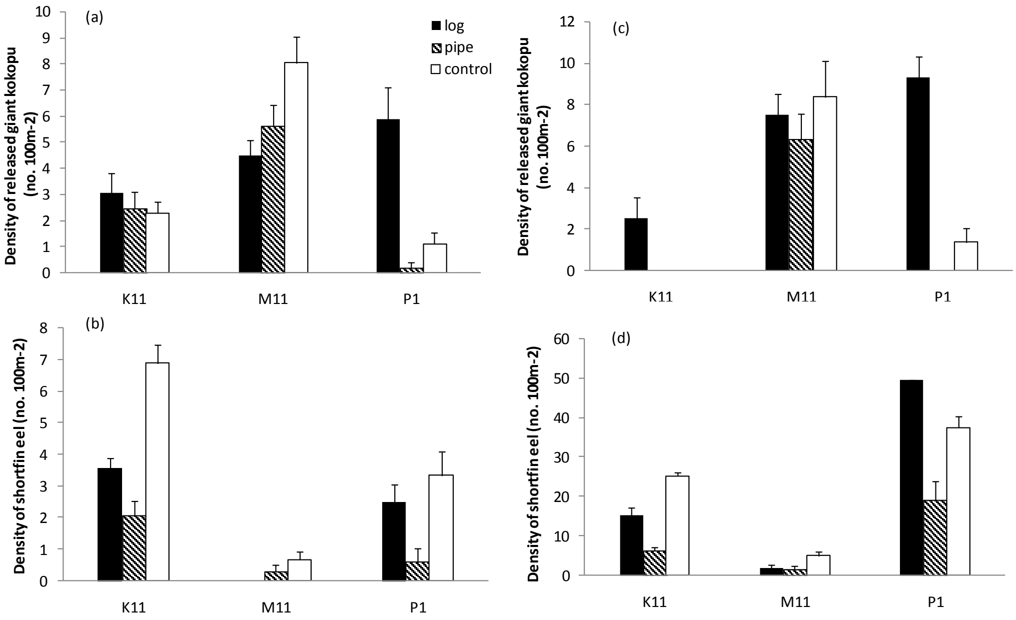

Figure 17: Mean density (no. 100 m^{-2} ± 95% CI) (average of all dates) of (a) released giant kokopu observed spotlighting, (b) shortfin eel observed spotlighting, (c) released giant kokopu captured electric fishing and (d) shortfin eel captured electric fishing, in restored 20-m sections of habitat at three sites (K11, M11 and P1). Spotlighting was conducted monthly (4/4/07 to 21/11/07) and electric fishing twice (June 07 and November 07) following habitat enhancement (Note: y axes vary).

4.3.5.2 Relationship with wild giant kokopu

Wild and stocked giant kokopu were compared within the five stocked streams to determine any relationships between farm-reared fish and wild fish abundance. Three of the five sites had wild giant kokopu present initially (Tables 11 and 12), and all sites except M11 had wild giant kokopu present at some stage during the study. Both wild and stocked fish were found to co-occur at Site P1 (Table 11 and 12). Wild and stocked giant kokopu were found to co-occur in winter at Site K11, with higher densities and lower biomass indicating the occurrence of juvenile giant kokopu which may have been a result of recruitment, although giant kokopu were not recaptured after July at K11. No stocked giant kokopu were recaptured while electric fishing at Site K10b, however, wild giant kokopu were captured on each sampling occasion (Table 11). A greater number of wild giant kokopu were caught using trap nets than electric fishing due to the greater frequency of trap netting (Table 12). Both fishing methods indicate similar results.

Table 11: Released and wild giant kokopu caught using electric fishing at the five release sites (a) total density (no. 100 m^{-2}) and (b) total biomass (g m^{-2}).

(a) Density (no. 100 m^{-2})

Season	Giant kokopu	K10a	K10b	K11	P1	M11
Summer 06	Stocked	0.0	0.0	0.0	0.0	0.0
	Wild	3.8	4.5	0.0	5.9	0.0
Winter 07	Stocked	0.0	0.0	5.1	15.2	23.9
	Wild	0.0	4.9	13.0	12.4	0.0
Summer 07	Stocked	0.0	0.0	0.0	6.2	20.5
	Wild	0.0	4.6	0.0	3.1	0

(b) Biomass (g m^{-2})

Season	Giant kokopu	K10a	K10b	K11	P1	M11
Summer 06	Stocked	0.00	0.00	0.00	0.00	0.00
	Wild	12.51	1.71	0.00	9.16	0.00
Winter 07	Stocked	0.00	0.00	1.36	0.77	0.84
	Wild	0.00	12.52	0.32	0.77	0.00
Summer 07	Stocked	0.00	0.00	0.00	1.30	3.56
	Wild	0.00	12.38	0.00	2.17	0

Table 12: Released and wild giant kokopu caught using trap nets at five release sites, (a) density (no. 100 m^{-2}) and (b) biomass (g m^{-2}). Five Gee minnow nets were set at all sites, and fyke nets at sites K10b (1), K10a (3) and M12 (3).

a. Density (no. 100 m^{-2}) by trap nets

Season	Giant kokopu	K10a	K10b	K11	P1	M11
Initial	Stocked	0.0	0.0	0.0	0.0	0.0
	Wild	4.2	0.0	4.6	0.0	0.0
April	Stocked	0.0	0.0	0.3	4.8	3.8
	Wild	1.9	5.0	25.0	0.0	0.0
May	Stocked	0.0	0.0	0.0	6.2	6.9
	Wild	0.0	0.0	25.5	0.0	0.0
June	Stocked	0.0	5.0	4.2	0.0	7.1
	Wild	2.2	5.0	0.0	0.0	0.0
July	Stocked	0.0	0.0	0.0	3.1	17.4
	Wild	0.0	5.0	0.0	3.1	0.0
August	Stocked	0.0	4.5	0.0	0.0	18.0
	Wild	0.0	5.0	0.0	0.0	0.0
September	Stocked	0.0	0.0	0.0	0.0	9.9
	Wild	4.4	0.0	0.0	0.0	0.0
October	Stocked	0.0	0.0	0.0	0.0	13.4
	Wild	0.0	5.0	0.0	0.0	0.0
November	Stocked	0.0	0.0	0.0	3.1	20.6
	Wild	0.0	0.0	0.0	0.0	0.0

b. Biomass (g m^{-2}) by trap nets

Season	Giant kokopu	K10a	K10b	K11	P1	M11
Initial	Stocked	0.00	0.00	0.00	0.00	0.00
	Wild	14.02	0.00	0.03	0.00	0.00
April	Stocked	0.00	0.00	0.38	0.15	0.11
	Wild	4.41	21.19	0.88	0.00	0.00
May	Stocked	0.00	0.00	0.00	0.21	0.28
	Wild	0.00	0.00	1.70	0.00	0.00
June	Stocked	0.00	0.15	0.41	0.00	0.27
	Wild	3.78	2.61	0.00	0.00	0.00
July	Stocked	0.00	0.00	0.00	0.35	0.99
	Wild	0.00	0.25	0.00	0.24	0.00
August	Stocked	0.00	0.23	0.00	0.00	1.19
	Wild	0.00	3.94	0.00	0.00	0.00
September	Stocked	0.00	0.00	0.00	0.00	1.22
	Wild	12.50	0.00	0.00	0.00	0.00
October	Stocked	0.00	0.00	0.00	0.00	1.85
	Wild	0.00	4.09	0.00	0.00	0.00
November	Stocked	0.00	0.00	0.00	0.84	2.89
	Wild	0.00	0.00	0.00	0.00	0.00

4.4 Discussion

The juvenile giant kokopu released into Hamilton urban streams were farm-reared and considered naïve of natural environmental variability and abiotic interactions. Their reaction to release and habitat enhancement was of interest because of the potential to use farm-reared fish in stream restoration.

Fish were released into five sites and had the greatest retention rate at Site M11. This site had several distinctive features: (1) smallest of the sites, (2) overhanging vegetative buffer, (3) existing pool, riffle and run habitat, (4) clear water (although iron floc common) and (5) low biomass and density of predatory shortfin eels. Fish were recaptured at sites K11 and P1 which were larger but nevertheless shallow and clear; one fish was caught at Site K10b which was deep, while no fish were recaptured at the deepest and widest Site K10a. K10b and K10a were both deep turbid streams, which may have affected fish retention or the efficiency of recapture methods. Water clarity has been found to effect migrating juvenile *Galaxias* species in tank and field trials. Banded kokopu are known to strongly avoid suspended sediment in tank trials, while inanga and koaro showed some avoidance (Boubee et al., 1997). Richardson et al. (2001) also found that juvenile migratory banded kokopu preferred clear water, showing significantly less upstream migration in turbid waters in field trials. Turbidity may play a similar role in influencing juvenile giant kokopu distribution.

Three fishing methods were used at Site M11 to assess the released fish population. Spotlighting was found to be the best method for assessing the fish population because; habitat use could also be assessed, it was relatively non-invasive, and fish could be captured. Trap nets indicated a misleading increase in fish densities over time. Electric fishing frequency was limited because of the possibility of habitat disturbance, adverse effects on fish, and the effort required (Mesa and Schreck, 1989). Spotlighting has been found to yield similar population estimates to electric fishing for banded kokopu and is a preferred method in small streams as it is less labour intensive and less invasive (McCullough and Hicks, 2002). There was a decrease in the number of fish observed during spotlighting over the winter months. This decrease could be due to seasonal changes in giant kokopu behaviour due to cooler water temperatures. Salmonid behaviour is known to decrease in winter as a result of lower temperatures and this was also found with giant kokopu in Otago streams (David and Closs, 2003).

Fish released into streams varied in size from 42 to 78 mm in length (measured when marked), which was similar to the migratory stage of 40-55 mm (McDowall and Kelly, 1999). Only 17 of the fish were less than 55 mm, 13 of these were from P1, where fish were held in the lab longer than the other sites, and were therefore likely to have grown prior to release. Fish were held for three days prior to release at sites K10a, K10b and K11 and six days prior to release at Site K11. Fish >50 mm were likely to remain in release reaches and select suitable habitat because they were likely to be past the migrant stage (McDowall and Kelly, 1999).

The juvenile giant kokopu showed a limited and variable response to the habitat enhancement trials, with faster growth in the log section of the five streams assessed. This could reflect increased food supply within accumulated organic matter associated with logs (see Chapter 3) or the ability to use log structures to provide refugia from predation and high flows. Bonnett et al. (2002) restored habitat using edge logs which provided cover, and this proved to be effective for adult giant kokopu in streams larger than Site M11. Mid-channel logs used in this study appeared to be effective in small shallow streams such as Site M11 (but not larger streams such as K10a) at providing habitat for juvenile giant kokopu, suggesting that edge logs could be a better option for larger urban streams.

Despite the slight preference for the log section over the pipe section the overall preference for existing habitat seemed to outweigh the habitat provided by enhancement, particularly at Site M11. At Site M11 fish seemed to prefer the control section, which had some undercut bank habitat, pools and possible refuge between rocks in the gabion basket on the true left side of the stream. Later in the study, high winter flows caused sediment movement forming pools behind the ponga logs in the log treatment of this stream. By definition this would be suitable habitat, but was not utilised by giant kokopu, which appeared to display strong fidelity to initial site selection. It is possible that juvenile giant kokopu did not discover this habitat as these fish are known to inhabit short predictable home ranges (David and Closs, 2003). Although these authors studied adult giant kokopu, juveniles at M11 appeared to have limited home ranges as recaptures at the same locations seemed common.

Juvenile giant kokopu were not captured at Site K11 after July, which coincided with winter high flow events. David and Closs (2002) found that fish had the ability to move and settle elsewhere during high flows (which may have been the case at K11), stay within their home range, or move and return.

The habitat preference of juvenile giant kokopu may differ from adult habitat, but is not well defined, as adults are captured more frequently in field studies (Bonnett and Sykes, 2002). Ecological requirements of fish may change with size, resulting in different habitat selection, as shown by Whitehead et al. (2002), who found that small giant kokopu appeared to select quite different habitat to giant kokopu in the medium and large size classes. Juveniles were most commonly found in water with a depth less than 0.1 m, whilst larger fish were found in deeper water. Juvenile fish were commonly encountered in faster flowing backwaters, while larger fish were typically seen in slower flowing pools while spotlighting (Whitehead et al., 2002).

At Site M11 where individual fish were observed to reside within the same pool, one fish was usually substantially larger than the other fish, suggesting hierarchical dominance. David and Stoffels (2003) found that the heaviest individual attained dominance within a pool, this position was usually at the head of the pool where drift-feeding fish have a better feeding advantage. Juvenile giant kokopu also exhibited hierarchical behaviour where densities of adult giant kokopu were low (David and Stoffels, 2003).

Growth of the juvenile giant kokopu in Hamilton urban streams varied markedly between individual sites. Growth of fish is influenced by a number of biotic and abiotic factors. Interestingly, the growth rates of these fish were quite substantial in winter when water temperatures were low. Fish growth is related to fish abundance and food availability, whereby abundance increases with food availability (West et al., 2005). Urban streams are typically degraded waterways and as such support a degraded macroinvertebrate community; oligochaetes were observed to be the most common invertebrate species on drift nets while electric fishing. Terrestrial food sources are known to be an important component of giant kokopu diet where riparian vegetation is present (Bonnett and Lambert, 2002). Growth rate is significantly greater for smaller fish (Allibone et al., 2003). Fish were not sexed during this research but female banded kokopu were found to grow faster in samples from the South Island and Owhiro Stream, Wellington (Hopkins, 1979). Ontogenetic differences may account for some of the variability in growth rates in the present study. West et al. (2005) found annual growth rates for banded kokopu of 150 mm tail length ranged from 3.6 to 15.6 mm, while Hopkins (1979) found a growth rate of 12.5 to 30.0 mm per year for banded kokopu. Juvenile giant kokopu were smaller than this size during this research and had variable growth, ranging from 11.0 to 40.2 mm per year (extrapolated estimate, as fish were not sampled over a full year), which is similar to rates found by Hopkins (1979).

Released giant kokopu were recaptured most often at Site M11 where large (potentially piscivorous) eels were uncommon. Migrating juvenile inanga appeared to actively avoid shortfin eel odours in tank trials (McLean et al., 2007). This may be due to innate avoidance behaviour to a natural predator as the giant kokopu had not encountered shortfin eels in their farmed environment.

Studies have shown that juvenile *Galaxias* species have a positive response to the odour of conspecifics. The present study did not indicate a positive relationship between wild giant kokopu and juvenile retention. However, results were limited by the low number of adult giant kokopu present at release sites and complicating factors of a natural environment. In laboratory tank trials, Baker and Hicks (2003) found that juvenile inanga were attracted to adult inanga, banded kokopu and koaro odours. Baker and Montgomery (2001) have shown that juvenile migratory banded kokopu had the ability to discriminate species-specific pheromones which influence habitat selection. These authors also found that at high odour concentrations, juvenile migration response was retarded. Odour of conspecifics appears to be integral in habitat selection as it can override avoidance response to unsuitable habitat (such as high suspended solid concentrations) (Baker, 2003). These results suggest that positive attraction to odour was not a factor influencing retention of released fish, although it may influence natural recruitment.

Chapter 5 Summary and recommendations

5.1 Restoration of fish habitat in urban streams

Habitat enhancement and stocking of farm-reared juvenile giant kokopu were used to determine if Hamilton urban streams were habitat or recruitment limited, arising from a lack of suitable habitat or limited access, respectively. The limited response to enhancement structures suggested that fish were not habitat limited or, that the structures used were ineffective (e.g., larger logs may have provided better habitat). The survival and retention of released farm-reared giant kokopu suggest that Hamilton urban streams may be partially recruitment limited, and that narrow, clear, shallow streams (such as M11) offer better habitat suitability than wide, deep, turbid streams (such as K10a). The small population size of wild giant kokopu suggests that barriers to migration, distance to the ocean, or low numbers of natural recruits are limiting wild populations. Habitat restoration will have limited value if fish cannot gain access to restored sections, and steps to overcome this should be considered before releasing or undertaking habitat enhancement in appropriate streams.

Potential habitats were enhanced using hollow clay pipes and ponga logs, and structural stability and microhabitat changes around the structures were monitored. Monitoring of the structures revealed that pipes were less stable and were observed to infill faster, making them less effective than logs at providing habitat. Structure and sediment movement appeared to correlate with high intensity rainfall events, caused by peak flows as a result of impervious cover (Paul and Meyer, 2001; Wang et al., 2001; Herricks and Schwartz, 2004; Roy et al., 2005). Microhabitat alterations were greater around the log structures than the pipes, with more CPOM (coarse particulate organic matter) retention, scour and deposition, although differences were small. CPOM is an important component of physical habitat because it can provide refugia for fish as well as a possible food source through macroinvertebrate colonisation (Richardson and Neill, 1991).

Three fishing methods were used to study fish populations during this study. Electric fishing was conducted three times throughout the duration of the research, once prior to habitat enhancement, again in June 2007 and November 2007. The frequency of electric fishing was limited because it was potentially detrimental to fish and may have disturbed habitat (Mesa and Schreck, 1989). Electric fishing resulted in the capture of higher densities of shortfin eel than spotlighting, whereas spotlighting appeared to yield similar capture results to electric fishing for released juvenile giant kokopu. Unbaited trap nets were set monthly at all 10 sites and provided seasonal information on fish population changes within treatments. Traps yielded lower biomass and densities over winter, presumably reflecting fish being less active (David and Closs, 2003) with recovery in summer, showing the considerable temperature dependence of this method. Although disruptive, electric fishing was considered to be the best method to monitor wild fish populations because of the limitations of water clarity for spotlighting, and low winter capture rates using trap nets.

A greater number of fish including wild giant kokopu and mosquitofish were caught in the log section of streams, resulting in higher biomass and densities; these were not statistically significant by ANOVA because of the complex array of interactions, and the small number of replicates. Koura appeared to prefer the log sections, but this preference was also statistically insignificant by ANOVA. Smelt recruited into streams with good access in winter 2007. Whitebait recruited into urban streams from November in both summer 2006 and 2007. Habitat enhancement using logs was considered to be partially successful in the short term. Up-scaling the size of the logs used to enhance habitat, or the length of the reach manipulated, may provide better overall habitat for fish species.

The released giant kokopu did not respond significantly to habitat enhancement; however growth may be faster in log sections, possibly reflecting greater food production. Released fish were observed and captured at approximately the same localities on each occasion, suggesting fidelity to initial habitat selection may have driven juvenile giant kokopu distribution within stream sections. Habitat considered suitable for giant kokopu, characterised by pools with small riffles and slow water velocity (Bonnett and Sykes, 2002; Whitehead et al., 2002), was created behind logs after sediment movement during winter months at Site M11, but was not utilised. The juvenile giant kokopu were retained in higher numbers, for a longer period of time, in streams that were: narrow, shallow, clear, had existing pool and riffle habitat, overhead cover, and low densities and biomass of eels. Released fish remained at three of the five release sites for at least eight months (the duration of monitoring) although subsequent observations indicate that fish remained after 11 months at Site M11. The retention time of the juvenile giant kokopu in urban streams may also be influenced by floods and available refugia. Released fish had substantial growth over winter months. Length growth was greatest at Site K11 where fish were not recaptured after July.

The desired result of restoring fish populations is to create suitable habitat for juveniles and adults of the target species. Habitat requirements can differ among life-stages, for example: juvenile giant kokopu prefer faster flowing backwaters, while adults reside in pools next to faster flowing water, indicating habitat needs of different life stages should be considered. At Site M11 where the stream was shallow, logs emerged slightly from the water surface creating a riffle effect with low water velocity behind the log. However, this slower flowing area may be more inclined to infill with the high sediment transport in urban streams. If pool habitat had been established and maintained hydraulically behind the log structures, an ideal habitat for giant kokopu could have been created.

Shortfin eels were the dominant species captured in Hamilton urban streams; densities were more variable but similar to those found in pasture streams but biomass could be considerably higher (Hicks and McCaughan, 1997; Hicks, 2003). Juvenile giant kokopu were retained in highest numbers where eel biomass was lowest, but overall there was no significant relationship. There appeared to be no relationship between the retention of released giant kokopu and the presence of wild giant kokopu in Hamilton urban streams, suggesting that existing populations are not required for the establishment of new populations using re-introduction. This study shows that naïve farm-reared giant kokopu can grow rapidly and survive for at least 11 months when introduced to suitable urban streams, although after time, as fish grow, they may need to seek out other suitable habitat (e.g. larger pools). The results of this study indicate that potential exists for the use of farm-reared individuals in stream restoration.

5.2 Recommendations for the use of farm-reared giant kokopu

Transported juvenile giant kokopu experienced high mortality shortly after arrival to the University of Waikato laboratory as a result of movement stress, high water temperatures and existing parasites and diseases. To minimise stress, it is recommended that fish are only transported once, from the farm-rearing holding tank, to the release site. Familiarising or training of naïve farm-reared fish (e.g., to eel odour) in tank trails may increase survival rate following release. However, untrained farm reared fish had a good reaction to release where conditions were ideal, and training may not increase survival. Released fish survival was greatest where potentially piscivorous eels were removed, therefore eel removal prior to release and throughout population establishment should be considered to aid survival.

Recruitment into streams is important for establishment and retention of a fish population. Giant kokopu are diadromous fish and as such undergo migration between the ocean and freshwater. Barriers to stream migration can limit recruitment and therefore fish populations. In the long term, restoration initiatives need to ensure fish recruitment can be sustained in the form of fish passage. Rehabilitating migration paths or active introduction can overcome recruitment limitations.

Log structures can alter in-stream microhabitat resulting in habitat complexity, which may act as refugia for juvenile giant kokopu from predation as well as high flows. Fish should be released after suitable habitat has been established because fish seem to exhibit initial habitat selection fidelity. For example, habitat within the log section of Site M11 appeared to be ideal for giant kokopu after winter sediment movement, but was not utilised by fish which exhibited limited movement, suggesting that this habitat may not have been discovered.

If water clarity permits spotlighting should be used as the method for monitoring nocturnal giant kokopu populations. Spotlighting is ideal because fish can be caught with relative ease, in shallow and narrow streams, and their behaviour and to some extent habitat use can be observed. Spotlighting is also less labour intensive and less invasive on fish than either trap netting or electric fishing methods.

5.3 Recommendations for future work

Marking/tagging of individual fish would have provided data on individual growth, residence and habitat use. Tracking fish could be used to determine habitat requirements and movement of fish in degraded urban environments; in particular movement during high flows would be of interest. However, radio tracking is usually restricted to adult fish due to the size of the device. Adult giant kokopu were tracked in Cullen's and Alex Creek, South Island, New Zealand, by radio-telemetry, to determine seasonal variation in microhabitat use and behaviour. The use of radio-telemetry on adult giant kokopu provided insight into home range use and flood movements (in natural streams) (David and Closs, 2003).

The restored sections of stream were relatively short, and restoring areas connecting suitable habitats may have provided the fish with better overall habitat. Connected restoration of spawning, juvenile and adult habitats is needed for population recovery (Lake et al., 2007). As well as longer reaches, bigger logs may have provided better habitat, especially in larger streams, where the size of the structure should be a function of stream size. Side logs may provide better habitat for fish, including giant kokopu, in larger streams.

References

Aldridge, B. M. T. A., Hicks, B. J. 2006. The distribution of fish in the urban gully system streams of Hamilton City. *CBER Contract Report No. 48.* Client report prepared for Environment Waikato and Hamilton City Council. Centre for Biodiversity and Ecology Research, Department of Biological Sciences, School of Science and Engineering, The University of Waikato, Hamilton.

Allibone, R., Caskey, D., Miller, R. 2003. Population structure, individual movement, and growth rate of shortjaw kokopu (*Galaxias postvectis*) in two North Island, New Zealand streams. *New Zealand Journal of Marine and Freshwater Research 37:* 473-483

Baker, C. F. 2003. Effect of adult pheromones on the avoidance of suspended sediment by migratory banded kokopu juveniles. *Journal of Fish Biology 62:* 386-394

Baker, C. F., Hicks, B. J. 2003. Attraction of inanga (*Galaxias maculatus*) and koaro (*Galaxias brevipinnis*) juveniles to adult galaxiid odours. *New Zealand Journal of Marine and Freshwater Research 37:* 291-299

Baker, C.F., Montgomery, J. C. 2001. Species-specific attraction of migratory banded kokopu juveniles to adult pheromones. *Journal of Fish Biology 58*: 1221-1229

Baker, C. F., Smith, J. P. 2007. Habitat use by banded kokopu (*Galaxias fasciatus*) and giant kokopu (*G. argenteus*) co-occurring in streams draining the Hakarimata Ranges, New Zealand. *New Zealand Journal of Marine and Freshwater Research 41:* 25-33

Beard, S. 2007a. Regional River Water Quality Monitoring Programme: Data Report 2006. Environment Waikato technical report 2007/12. Environment Waikato, Hamilton East, New Zealand.

Beard, S. 2007b. Waikato River Water Quality Monitoring Programme: Data Report 2006. Environment Waikato technical report 2007/18. Environment Waikato, Hamilton East, New Zealand.

Bernhardt, E. S., Palmer, M. A. 2007. Restoring streams in an urbanizing world. Freshwater Biology 52, 738-751

Bonnett, M. L., Lambert, P. W. 2002. Diet of giant kokopu, *Galaxias argenteus*. *New Zealand Journal of Marine and Freshwater Research 36:* 361-369

Bonnett, M. L., Sykes, J. R. E. 2002. Habitat preference of giant kokopu, *Galaxias argenteus*. *New Zealand Journal of Marine and Freshwater Research 36:* 13-24

Bonnett, M. L., McDowall, R. M., Sykes, J. R. E. 2002. Critical habitats for the conservation of giant kokopu, *Galaxias argenteus* (Gmelin, 1789). *Science for Conservation 206.* 50p.

Borg, D., Rutherford, I., Stewardson, M. 2007. The geomorphic and ecological effectiveness of habitat rehabilitation works: Continuous measurement of scour and fill around logs in sand-bed streams. *Geomorphology 89*: 205-216

Boubee, J. A. T., Dean, T. L., West, D. W., Barrier, R. F. G. 1997. Avoidance of suspended sediment by the juvenile migratory stage of six New Zealand native fish species. *New Zealand Journal of Marine and Freshwater Research 31*: 61-69

Brown, K. B. 2000. Urban Stream Restoration Practices: An Initial Assessment. The Center for Watershed Protection. Elliot City, MD.

Cassin, J. L., Tear, L., Fuerstenberg, R., Whiting, D., St. John, D., Murray, B. 2004. Sustainable river restoration in urban streams – using biological indicators to establish environmental flow targets in the Pacific Northwest. *Fifth International Symposium on Ecohydraulics. Aquatic Habitats: Analysis and Restoration. Madrid*

Chapman, M. A. 1996. Human impacts on the Waikato River system, New Zealand. *GeoJournal 40*: 85-99

Clarkson, B. D.; McQueen, J. C. 2004. Ecological restoration in Hamilton City, North Island, New Zealand. *Society for Ecological Restoration.* Victoria, Canada.

Collier, K. J., Cooper, A. B., Davies-Colley, R. J., Rutherford, J. C., Smith, C. M., Williamson, R. B. 1995. Managing Riparian Zones: A contribution to protecting New Zealand’s rivers and streams. Department of Conservation. Wellington, New Zealand.

Collier, K. J., Kelly, J. 2005. Regional guidelines for ecological assessments of freshwater environments: macroinvertebrate sampling in wadeable streams. Environment Waikato Technical Report 2005/02. Environment Waikato, Hamilton East, New Zealand.

Cordova, J. M., Rosi-Marshall, E. J., Yamamuro, A. M., Lamberti, G. A. 2007. Quantity, controls and functions of large woody debris in midwestern USA streams. *River Research and Applications 23*: 21-33

Crook D. A., Robertson, A. I. 1999. Relationships between riverine fish and woody debris: implications for lowland rivers. *Marine and Freshwater Research 50:* 941-953

David, B. 2002. Threatened fishes of the world: *Galaxias argenteus* (Gmelin, 1769) (Galaxiidae). Environmental Biology of Fishes 63: 416

David, B. O., Closs, G. P. 2002. Behaviour of a stream-dwelling fish before, during, and after high-discharge events. *Transactions of the American Fisheries Society 131:* 762-771

David, B. O., Closs, G. P. 2003. Seasonal variation in diel activity and microhabitat use of an endemic New Zealand stream-dwelling galaxiid fish. *Freshwater Biology 48:* 1765-1781

David, B. O., Stoffels, R. J. 2003. Spatial organisation and behavioural interaction of giant kokopu (*Galaxias argenteus*) in two stream pools differing in fish density. *New Zealand Journal of Marine and Freshwater Research 37:* 315-322

David, B., Chadderton, L., Closs, G., Barry, B., Markwitz, A. 2004. Evidence of flexible recruitment strategies in coastal populations of giant kokopu

(*Galaxias argenteus*). DOC Science Internal Series 160. Department of Conservation, Wellington, New Zealand.

De Jong, M. C. V., Cowx, I. G., Scruton, D. A. 1997. An evaluation of in-stream habitat restoration techniques on salmonid populations in a Newfoundland stream. *Regulated Rivers-Research and Management 13*: 603-614

Ebersole, J. L., Liss, W. J., Frissel, C. A. 1997. Restoration of stream habitats in western United States: Restoration as Reexpression of Habitat Capacity. *Environmental Management 21*: 1-14

Evans, B. F., Townsend, C. R., Crowl, T. A. 1993. Distribution and abundance of coarse woody debris in some southern New Zealand streams from contrasting forest catchments. *New Zealand Journal of Marine and Freshwater Research 27:* 227-239

Farahmand, T., Fleming, S. W., Quilty, E. J. 2007. Detection and visualization of storm hydrograph changes under urbanization: An impulse response approach. *Journal of Environmental Management 85:* 93-100

Fuselier, L. Edds, D. 1995. An artificial riffle as restored habitat for the threatened Neosho Madtom. *North American Journal of Fisheries Management 15:* 499-503

Glova, G. J., Bonnett, M. L., Docherty, C. R. 1985. Comparison of fish populations in riffles of three braided rivers in Canterbury, New Zealand. *New Zealand Journal of Marine and Freshwater Research 19:* 157-165

Gomi, T., Sidle, R. C., Bryant, M. D., Woodsmith, R. D. 2001. The characteristics of woody debris and sediment distribution in headwater streams, south eastern Alaska. *Canadian Journal of Forest Research 31*: 1386-1399

Herricks, E. E., Schwartz, J. S. 2004. River naturalization: habitat and ecohydraulics in an urban stream. *Fifth International Symposium on Ecohydraulics. Aquatic Habitats: Analysis and Restoration. Madrid.*

Hicks, B. J. 2003. Distribution and abundance of fish and crayfish in a Waikato stream in relation to basin area. *New Zealand Journal of Zoology 30*: 149-160

Hicks, B. J., McCaughan, H. M. 1997. Land use, associated eel production, and abundance of fish and crayfish in streams in Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research 31:* 635-650

Hopkins, C. L. 1979. Age-related growth characteristics of *Galaxias fasciatus* (Salmoniformes: Galaxiidae). *New Zealand Journal of Marine and Freshwater Research 13*: 39-46

House, R. A., Boehne, P. L. 1985. Evaluation of in-stream enhancement structures for salmonid spawning and rearing in a coastal Oregon stream. *North American Journal of Fisheries Management 5:* 283-295

House, R. A., Boehne, P. L. 1986. Effects of in-stream strutures on salmonid habitat and population in Tobe Creek, Oregon. *North American Journal of Fisheries Management 6:* 38-46

Jellyman, D. J. 1979. Observations on the biology of the giant kokopu, *Galaxias argenteus* (Gmelin, 1789). *Mauri Ora 7*: 53-61

Jowett, I. G., Richardson, J. 1996. Distribution and abundance of freshwater fish in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research 30:* 239-255

Jowett, I. G., Richardson, J. 2003. Fish communities in New Zealand rivers and their relationship to environmental variables. *New Zealand Journal of Marine and Freshwater Research 37*: 347-366

Jowett, I. G., Parkyn, S. M., Richardson, J. 2008. Habitat characteristics of crayfish (*Paranephrops planifrons*) in New Zealand streams using generalised additive models (GAMs). *Hydrobiologia 596:* 353-365

Kail, J., Hering, D. 2005. Using large wood to restore streams in Central Europe: potential use and likely effects. *Landscape ecology 20*: 755-772

Lake, P. S., Bond, N., Reich, P. 2007. Linking ecological theory with stream restoration. *Freshwater Biology 52*: 597-615

McCraw, J.D. 2000. Geology of Hamilton gullies. Pages 508 in Clarkson, B.D., McGowan, R., and Downs, T.M., editors. Hamilton Gullies – A workshop hosted by The University of Waikato and sponsored by the Hamilton City Council, 29-30 April 2000, Hamilton. The University of Waikato.

MsCraw, J. D. 2002. Physical environment: Botany of the Waikato. Waikato Botanical Society Inc., Department of Biological Sciences, The University of Waikato, Hamilton.

McCullough C. D., Hicks, B. J. 2002. Estimating the abundance of banded kokopu (*Galaxias fasciatus* Gray) in small streams by nocturnal counts under spotlight illumination. *New Zealand Natural Sciences 27*: 1-14

McDonald, A. E. 2007. Improving the success of a translocation of black mudfish (*Neochanna diversus*). MSc thesis, University of Waikato, Hamilton.

McDowall, R. M., Kelly G. R. 1999. Date and age at migration in juvenile giant kokopu, *Galaxias argenteus* (Gmelin) (Teleostei: Galaxiidae) and estimation of spawning season. *New Zealand Journal of Marine and Freshwater Research 33*: 263-270

McDowall, R. M. 2000. The Reed field guide to New Zealand freshwater fishes. Reed Publishing (NZ) Ltd, Auckland.

McLean, F., Barbee, N. C., Swearer, S. E. 2007. Avoidance of native versus non-native predator odours by migrating whitebait and juveniles of the common galaxiid, *Galaxias maculatus*. *New Zealand Journal of Marine and Freshwater Research 41:* 175-184

Mesa, M. G., Schreck, C. B. 1989. Electrofishing mark-recapture and depletion methodologies evoke behavioural and physiological changes in Cutthroat Trout. *American Fisheries Society 118*: 644-658

Minns, C. K. 199. Patterns of distribution and association of freshwater fish in New Zealand. *New Zealand Journal of Marine and Freshwater Research 24:* 31-44

Morgan, R. P., Cushman, S. F. 2005. Urbanization effects on stream fish assemblages in Maryland, USA. *Journal of North American Benthological Society 24*: 643-655

Mutz, M., 2004. Effect of wood addition on reach hydraulics and deposition of fine particulate organic matter in a degraded sand-bed stream. *Fifth International Symposium on Ecohydraulics. Aquatic Habitats: Analysis and Restoration. Madrid*

Negishi, J. N., Richardson, J. S. 2003. Responses of organic matter and macroinvertebrates to placements of boulder clusters in a small stream of Southwestern British Columbia, Canada. *Canadian Journal of Fisheries and Aquatic Sciences 60:* 247-258

Parkyn, S. M., Davies-Colley, R. J., Cooper, B. A., Stroud, M. J. 2005. Predictions of stream nutrients and sediment yield changes following restoration of forested riparian buffers. *Ecological Engineering 24*: 551-558

Paul, M. J., Meyer, J. L. 2001. Streams in the urban landscape. *Annual Review of Ecology and Systematics 32 :* 333-365

Piegay, H., Thevenet, A., Kondolf, G. M., Landon, N. 2000. Physical and human factors influencing potential fish habitat distribution along a mountain river, France. *Geografiska Annaler. Series A, Physical Geography 82:* 121-136

Richardson, J., Boubée, J. A. T., West, D. W. 1994. Thermal tolerance and preference of some native New Zealand freshwater fish. *New Zealand Journal of Marine and Freshwater Research 28:* 399-407

Richardson, J. S., Neill, W. E. 1991. Indirect effects of detritus manipulations in a montane stream. *Canadian Journal of Fisheries and Aquatic Sciences 48:* 776-783

Richardson, J., Rowe, D. K., Smith, J. P. 2001. Effects of turbidity on the migration of juvenile banded kokopu (*Galaxias fasciatus*) in a natural stream. *New Zealand Journal of Marine and Freshwater Research 35:* 191-196

Richardson, J., Taylor, M. J. (2002, reprinted with minor revisions 2004). A guide to restoring inanga habitat. *NIWA Science and Technology Series No. 50*. 31p

Roy, A. H., Freeman, M. C., Freeman, B. J., Wenger, S. J., Ensign, W. E., Meyer, J. L. 2005. Investigating hydrologic alteration as a mechanism of fish assemblage shifts in urbanizing streams. *Journal of North American Benthological Society 24:* 656-678

Scarsbrook, M. R., Townsend, C. R. 1994. The roles of grass leaf litter in streams draining tussock grasslands in New Zealand: retention, food supply and substrate stabilisation. *Freshwater Biology 32:* 429-444

Scott, J. B., Steward, C. R., Stober, Q. J. 1986. Effects of urban development on fish population dynamics in Kelsey Creek, Washington. *Transactions of the American Fisheries Society 115*: 555-567

Suren, A. M., McMurtrie, S. 2005. Assessing the effectiveness of enhancement activities in urban streams: II. Responses of invertebrate communities. *River Research and Applications 21:* 439-453

Tisdall, C. 1994. Setting priorities for the conservation of New Zealand's threatened plants and animals. Department of Conservation, Wellington. 64 pp.

Wall, K., Clarkson, B. 2001. Gully restoration guide; a guide to assist in the ecological restoration of Hamilton's gully systems. Hamilton City Council, Hamilton, New Zealand.

Wallace, J. B., Eggert, S. L., Meyer, J. L., Webster, J. R. 1997. Multiple trophic levels of a forest stream linked to terrestrial litter inputs. *Science 277:* 102-104

Wang, L., Lyons, J., Kanehl, P. 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. *Environmental Management 28:* 255-266

West, D. W., Jowett, I. G., Richardson, J. 2005. Growth, diet, movement, and abundance of adult banded kokopu (*Galaxias fasciatus*) in five Coromandel, New Zealand streams. *New Zealand Journal of Marine and Freshwater Research 39:* 915-929

Whitehead, A. L., David, B. O., Closs, G. P. 2002. Ontogenetic shift in nocturnal microhabitat selection by giant kokopu in a New Zealand stream. *Journal of Fish Biology 61:* 1373-1385

Wilding, T. K. 1998. The state of Hamilton streams. *Environment Waikato Technical Report 1997/17*

Wissmar, R. C., Beschta, R. L. 1998. Restoration and management of riparian ecosystems: a catchment perspective. *Freshwater Biology 40*: 571-585

Appendix 1

MINISTRY OF FISHERIES
Te Tautiaki i nga tini a Tangaroa

608 Rosebank Road, Avondale
PO Box 19747, Avondale
Auckland, New Zealand
Phone (09) 820-1990, Fax (09) 820

APPROVAL TO TRANSFER AND RELEASE LIVE AQUATIC LIFE
(between sites where the species already exists)
Transfer number AK108a

Pursuant to Section 26ZM(2) of the Conservation Act 1987, approval is hereby given to: -

Brenda Michelle Te Aroha Aldridge
% Waikato University
Private Bag 3105
HAMILTON

to transfer and release up to 200 giant kokopu (*Galaxias argenteus*) from a fish farm in Raglan to the following streams around Hamilton City:

- Mangakotukutuku Stream (four sites)
- Kirikiriroa Stream (five sites)
- Gibbons Creek, Parana Park (one site)

This approval is subject to the following conditions:

1 This approval is effective from the date of signature and expires on 30 April 2007. This approval replaces approval AK108, granted on 19 December 2006.

2 No more than 200 giant kokopu are to be released, and no more than 40 in any one stream.

3 Only farm-reared giant kokopu sourced from the Waitetuna River are to be released.

4 No fish taken from the wild are to be transferred.

5 All fish are to be treated for parasites and disease prior to being released.

6 This approval does not preclude any need that may exist for the holder to obtain necessary approvals from other agencies to release fish into these waters.

7 Records shall be kept of dates, quantities, and size range of giant kokopu released.

8 The Ministry of Fisheries must be advised when, where, and how many giant kokop were released.

D Lees

Daniel Lees
Aquaculture Manager
Nelson

21 / February. / 2007