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ABUNDANCE, BEHAVIOUR, AND HABITAT REQUIREMENTS OF THE BANDED KOKOPU

Galaxias fasciatus Gray (Pisces: Galaxiidae).

A thesis
submitted in partial fulfilment
of the requirements for the degree of
Master of Science (Zoology)
in
Biological Sciences
at the
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by
Cherie D. M^{c}Cullough B.Sc. (*JCUNQ*)

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1998

Frontispiece

Plate 0.01/. Kokopu at late ‘noon. Sawmill Stream, Coromandel.

ABSTRACT

The native freshwater banded kokopu (*Galaxias fasciatus* Gray) is generally regarded as a nocturnal predator of small native forest streams. Current methods for determining absolute abundances of this species in these streams is usually by the expensive, labour intensive, and invasive method of electric fishing. Spotlighting at night has previously been used to determine their presence and relative abundance, although the proportion seen (the bias of the method) was unknown. This bias was quantified by determining the numbers of banded kokopu seen at night in 20 m reaches against population estimates made in the same reaches the following day by removal electric fishing. This showed that, on average, only around 64% of fish of all sizes were being seen. Counts made by spotlighting therefore should be multiplied by 1.57 to correct for this bias and to estimate for the actual number of fish present. Spotlighting was found to be less efficient for smaller fish (approximately <60 mm total length), with only around 42% being observed. Spotlight counts for smaller fish therefore needed to be multiplied by 2.41 to estimate the actual abundances of fish of this size. Fish larger than this size, however, were more often seen (0.70) requiring their spotlight counts only to be multiplied by 1.42 to correct for their spotlight estimates.

Banded kokopu diel activity was studied using time-lapse video recordings with a low-light sensitive camera placed above pools in natural streams. Two major activity peaks were noted; one from 6:00 a.m. to around 8:00 a.m., the other from around 2:00 p.m. to around 7:30 p.m. Foraging behaviour, intra- and interspecific behaviours, and mode of swimming were all also noted.

Nocturnal habitat preferences and suitability functions were derived for all sizes of banded kokopu together, and then for the two aforementioned size classes separately. The following variables were tested for; mean column water velocity, surface water velocity, column depth, substrate size, and cover type. Used habitat data was collected at night by a colour filtered spotlight, with a small floating marker being placed at the focal point of the fish. The particular characteristics of this point were then measured along with available habitat data the following day. Fish were found to have a very narrow preference for very slow water velocities (around 0.05 m/s), with juveniles preferring slightly faster than adults when considered separately. Preferred water depths for both sizes were around 0.8 m, and preferred substrate sizes were fine gravels, sand, and silt. Preferred cover for all sizes was for tree and tree fern roots and undercut banks, as it also was for larger fish when considered separately. Smaller fish did not use tree fern roots significantly, but instead preferred the cover of cobbles. This smaller sized fish also made use of tree roots and undercut banks.

These new banded kokopu habitat preference data were then applied to an IFIM situation in the Waitakere River (West Auckland), where Resource Consent for abstraction of 0.0041 m^3/ s (total daily abstraction of 119 m^3) was being requested for irrigation of the adjacent Waitakere Golf Course. The headwaters of this river are situated in the pristine native forest of the Waitakere Ranges. This river also contains a significant dam (the Waitakere Reservoir) a few kilometres upstream of the proposed site of abstraction which diverts 16, 000 m^3 a day to out of stream uses. This figure represents approximately 80% of water that would otherwise continue downstream and eventually to the ecologically significant freshwater Te Henga Wetland. Changes in weighted useable area (WUA) as flow decreased as a result of this abstraction were modelled on the DOS program “RHYHABSIM” for fish and invertebrates known to be in this river. A comparison was also made of this habitat flow method with the North American Tennant method; an historical flow method which determines the effects of flows by set proportions of the mean low flow. No substantial reduction in WUA was noted for any of the species following abstraction from either a mean low flow (0.0200 m^3/s), or a one-in-five-year low flow (0.0134 m^3/s). Endorsement was therefore given for the proposed abstraction to Receive Consent. Recommendations on the time of day that the abstraction should take place, and other further methods to mitigate the ecological effects of this abstraction are also made.

PREFACE

"The Cock-a-bulla. This unholy looking fish is common all over the country wherever a bog hole or dark bush creek exists. ...I don't know what species of fish they belong to. The name is suggestive of bull trout, but it is more likely they are a species of their own, as from their appearance they have no business to be alive on earth now a days; their proper location is as a fossil in the coal formation or the old red sandstones.

That the Cock-a-bulla is a wicked fish there is not the slightest doubt as "they prefer the darkness rather than the light". They haunt box drains like evil spirits. If they can get down a well they are happy and in some mysterious way they do get down wells, and if one could only get into a pool down a coal pit the fish would have reached the height of Cock-a-bulla felicity."

Charles Douglas: Explorer (McDowall 1980)

Plate 0.02/. The endemic New Zealand banded kokopu *Galaxias fasciatus* (Gray).

THE BANDED KOKOPU *Galaxias fasciatus*

The banded kokopu *Galaxias fasciatus* Gray (order: Salmoniformes, family: Galaxiidae) is endemic to New Zealand and is found throughout the mainland and a few of its surrounding offshore islands (McDowall, 1990), and is our most common large galaxiid (Woods, 1963b; Graham, 1974). Although an especially strong climber, its inland penetration is usually low with population abundance typically being highest near the coast (West, 1989) (Fig. 0.01/.).

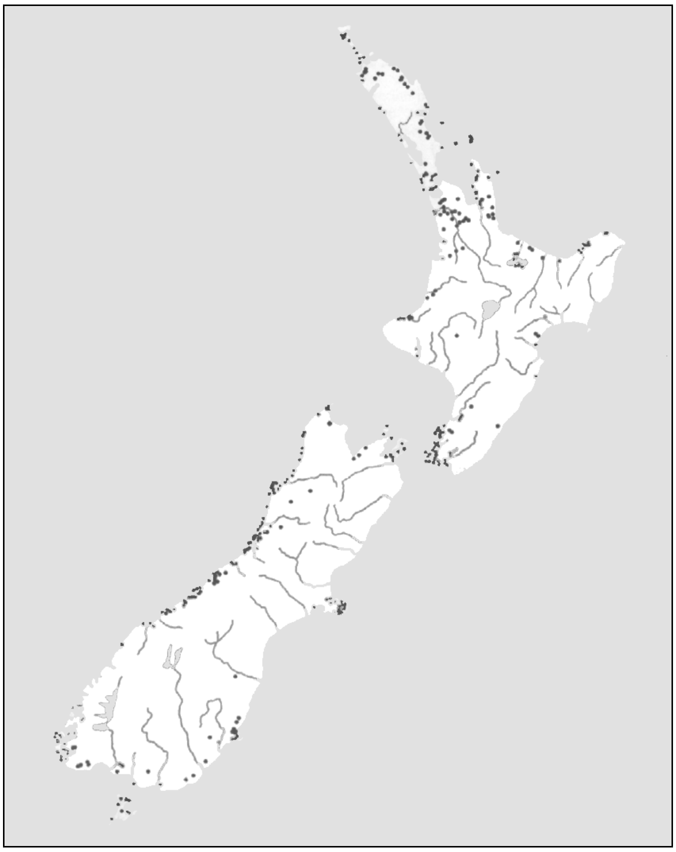

Figure 0.01/. National Freshwater Fish Database distribution of the indigenous banded kokopu on the New Zealand mainland and its offshore islands. From McDowall (1990).

Now usually known as banded kokopu, these fish were previously more commonly grouped by the early settlers with the other large galaxiids under the colloquial names “cock-abully”, “native-”, “Maori-”, or “mountain-” “trout” (Foord, 1990). Names historically used by the Maori (whom were particularly fond of this fish as a food item) (Phillipps, 1940) include “para”, kopakopako, and “kopu” among many others (Phillipps, 1947). It is, however, likely that these words were used for the other large native galaxiids also. One of the larger galaxiids of New Zealand with maximum total lengths to around 260 mm (McDowall, 1990), their typical habitat is in pools in slow flowing, small 1st-order headwater streams and tributaries with reasonably extensive riparian vegetation e.g. many characteristic small streams in native bush. Undercut banks, instream woody material and tree roots all appear to be strongly desired features of their habitat with this species rapidly becoming absent from streams when they begin to flow through deforested reaches e.g. streams leaving native forest and entering pastoral reaches below.

Plate 0.03/. Typical banded kokopu habitat with long slow pools and extensive over-hanging vegetation. Waipuna Stream (Hakarimata Ranges).

Most populations are diadromous (amphidromous sub-type), although lacustrine populations are also known in lakes and reservoirs both with and without streams allowing sea access. They are exclusively opportunistic predators, feeding largely on invertebrates of terrestrial origin falling from overhanging vegetation onto the waters surface (Main, 1988; Main & Lyon, 1988; Hanchet, 1990; McDowall, 1990; Swales & West, 1991; Halstead, 1994; Hicks, 1997). Shy and reclusive, they are often found close to urban areas when habitat is suitable, although observations of their existence are often simply limited to the noting of a disturbance when the occupied stream in question is approached. This elusiveness has lead to a fear of the state of the distribution and abundance of this publicly little recognised species. This concern has only begun to be allayed more recently with an increase in surveying with electric fishing equipment, and the establishment of such extensive record archiving as the New Zealand National Freshwater Fish Database (McDowall & Richardson, 1983). These data have indicated that whilst population abundance and distributions may be good in forested areas of the country, other areas with a paucity of tall woody riparian cover e.g. the anthropic deforestation of much of the historically forested Waikato region often exhibit a complete absence of their populations (West, 1989; Hicks & McCaughan, 1997).

A substantial contributor to the whitebait catch (McDowall & Eldon, 1980; Stancliffe *et al.,* 1988; Rowe *et al*., 1992), nationwide this species has also seen a decline in both the adults and their component proportions in these whitebait runs (McDowall, 1978; 1987; 1990). This fish is therefore currently considered to have a "threatened" conservation status (Swales & West, 1991; Tisdall, 1994), and is yet increasingly endangered through continued loss of adult habitat (McDowall, 1987; Swales, 1991) such as stream habitats in preferred native forest catchments by deforestation and other land-use changes.

> *"Rarity is the attribute of a vast number of species of all classes in all countries. If we ask ourselves why this or that species is rare, we can answer that something is unfavourable in its conditions of life: but what that something is we can hardly ever tell."*
>
> Darwin (1859)

Hopefully the research findings described in this dissertation will go some way toward explaining the decline of this special and unique part of New Zealand's all too little known endemic freshwater fish fauna.

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I gratefully acknowledge the assistance of the following people who helped with this thesis…

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Thanks must also be granted to my mother Lesley McCullough for nurturing both a love, empathy and respect for our awesome natural world; and also to my long-suffering partner Kylie Titchener who also somehow managed to put up with being everything from "emergency field technician and general packhorse", to secretary and yet is still around!

And finally, whilst not wanting to miss any of those whom also gave time and energy in the cause of this work; thanks to all the other people, too many to mention, who helped in some way to make this thesis happen. You know who you are…

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CHAPTER 1.

Determining the Abundances of Banded Kokopu in Streams by Spotlighting at Night.

"It is an extremely shy fish which moves very rapidly to cover when disturbed and so is seldom seen. In spite of this, the banded kokopu is easily observed at night with a flashlight, because the fish move freely in the stream pools at that time."

R. M. McDowall (1990), New Zealand freshwater fish: a guide and natural history.

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"*Because you know that night time, ooh…, is the right time…*"

Creedence Clearwater Revival (1969)

Plate 1.01/. Non-invasively measuring the total length of a large adult banded kokopu *in situ* under red light. Waipuna Stream, Hakarimata Ranges.

1.0 Introduction

Knowledge of abundances of a species is fundamental to their conservation management. As a consequence of the importance of these data, visual methods for determining the abundances of fishes in streams are not new. Techniques for estimating abundance of banded kokopu in their typical habitat of small streams have, however, been almost wholly limited to either the time-consuming method of electric-fishing, or to the use of small traps (e.g. Gee-minnow), and poisons (Hicks, 1985). The methods of overnight trap setting to obtain a population estimate via a declining removal of captures require many return trips to the stream, and poisoning is clearly not viable for almost all situations. Electric fishing has therefore remained the standard methodology for almost all wading depth stream surveying in New Zealand despite its high cost; potential risk to personnel; high labour and time consumption; and the possibility of injury, stress, or even death to individuals such as this highly stress sensitive species during capture (Neilson, 1996). Additionally, for a nocturnal species, which by very definition is more active at night, abundance estimates made during the day (as is required with electric fishing) clearly may be less efficient, and possibly even more inaccurate as a result of this low efficiency. This is most often seen with native galaxiids which may appear to be absent from a stream site in the day, yet only to appear abundant, and readily approachable (Plate 1.02/.) under torchlight that following night (McCullough, 1996b).

Figure 1.02/. Banded kokopu are so approachable under red light at night that they may even be stroked!

Drift-diving methods (so called "two-phase" techniques based on a double sampling design described by Cochran (1977)) are well established and recognised rapid methods of population estimation. These methods assess the size of fish populations more rapidly than the more expensive and labour intensive alternative of electric fishing and are typically targeted at salmonids. This rapid and less expensive method of population abundance determination is, however, not feasible to assess the populations of small stream dwelling New Zealand galaxiids. These fishes are notorious for being difficult and time-consuming to catch, such that regular and extensive analysis of population abundances are all too frequently not viable. Given this difficulty, however, methods for fish sampling in New Zealand to provide less resource consuming management tools for these problems have yet not evolved greatly in the last 30 years (McDowall, 1997b) with electric fishing remaining yet as a sole standard methodology. Furthermore, for these diurnally reclusive, nocturnally active species, the hours of darkness are therefore clearly the best times for abundance estimates to be made. An equivalent visual population estimation method to drift-diving, spotlighting is therefore proposed as an alternative to the grossly prevailing method of electric fishing for this species.

Spotlighting is not a new method for determining the presence of banded kokopu and other nocturnal native galaxiids in slow moving streams, with various authors noting the effectiveness of this method with this particular species (Main, 1988; McDowall, 1990). Indeed historically the Maori were known to capture banded kokopu in hand-nets under the light of flaming torches at night when they were found to be most approachable (Rowan Strickland, Cawthron Institute pers. comm.). What has, however, still remained undefined is the proportion of fish actually being seen i.e. the bias of the spotlighting estimate made was unknown.

The premise of the proposed spotlighting methodology as a tool for determining abundances of banded kokopu in streams is the same, and is based on those of the aforementioned established drift-diving methods. Methods which are not available in the shallow narrow waterways that banded kokopu typically inhabit. That is, given that there is a consistent relationship between actual measurements and visual observations i.e. "precision" is high enough (Fig. 1.01/.), then accuracy of estimate may be less important, as it is then possible to calculate a calibration ratio and "correct" for the bias that is associated with the visual observation technique (Hankin & Reeves, 1988).

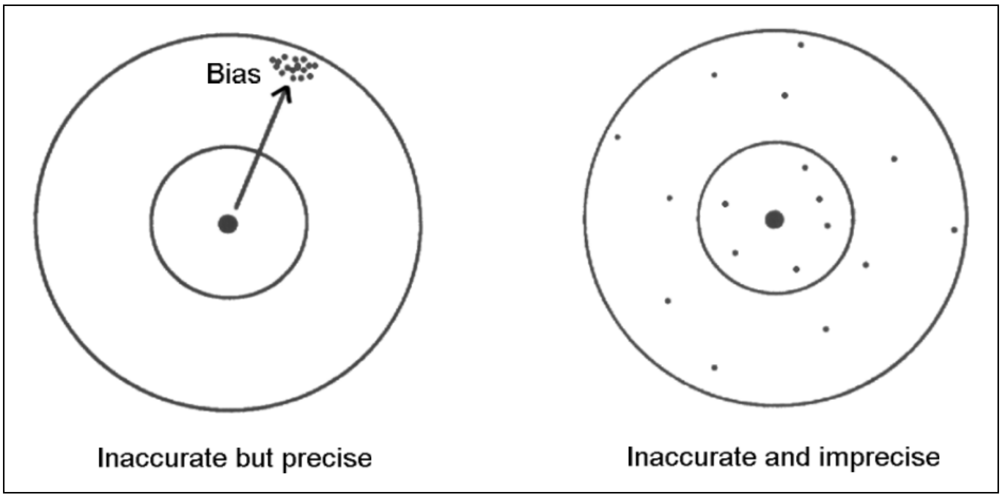

Figure 1.01/. The well used "target" analogy to explain the concept of accuracy, precision, and bias. After Jones & Marsh (1994).

When known, this bias corrector or "calibration factor" can then be used to correct for the total animals present, by a simply multiplication of the animals that were seen by the calibration factor.

1.1 Methods

1.11 Study sites

The streams sampled in both this study and also the further chapters, were in three major regions; the Coromandel Peninsula, West Auckland, and West Huntly (the Hakarimata Ranges) (Fig. 1.02/.). All sites were in native bush; either as mature canopy, or as well-established regeneration (Table 1.01/.). All streams also had high densities of banded kokopu, and were ≤ 2nd order in magnitude. Sampling took place within January to May 1997 with surveying Waipuna Stream occurring last, and the 2nd reach of Sawmill Stream being surveyed approximately 3 weeks before the other reaches of this stream. During these times water temperatures at the study sites were in the range from 12.2°C to 17.1°C.

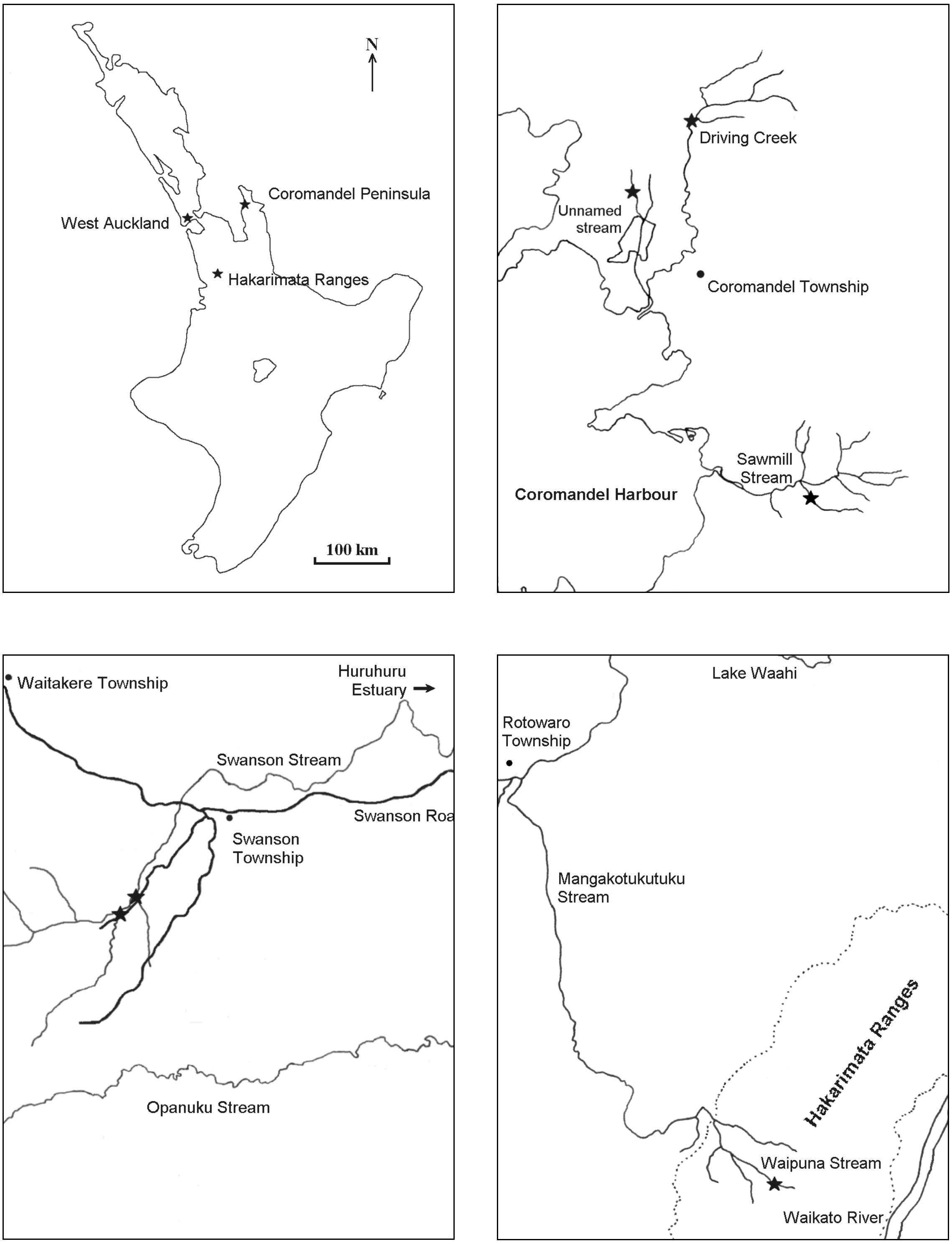

Figure 1.02/. Location of the three major study regions, and the study sites in these. Solid stars are sample regions and sites, solid circles are town locations (scale for all study site location maps = 1: 100 000).

Table 1.01/. Map coordinates and number of reaches sampled for each region and its streams.

Region	Stream name	Map coordinates / Altitude	Reaches sampled	Site description
Coromandel Peninsula	Driving Creek	T10 339.934 / c. 60 m	2	Fine substrate, modified old growth native forest
	Sawmill Stream	T11 357.871 / c. 30 m	4	Small cobble, largely unmodified catchment
	Unnamed stream	T10 328.922 / c. 25 m	2	Fine substrate, grazed undergrowth, regenerating
West Auckland	Swanson Stream	Q11 495.792 / c.60 m	6	Sandstone base, regenerating native forest
	Unnamed stream	Q11 498.795 / c.60 m	3	Sandstone base, regenerating native forest
Hakarimata Ranges	Waipuna Stream	S14 / c. 130 m	3	Hard greywacke base, pristine native forest

All the three Coromandel streams surveyed mixed with other tributaries and then flowed directly into the Coromandel Harbour. The two West Auckland tributaries sampled flowed into Swanson Stream, and then into the Waitemata Harbour. The Waipuna Stream was suspected to have mixed diadromous and lacustrine populations of banded kokopu given its long distance from the sea (approximately 100 km), and also that following mixing with the Mangakotukutuku Stream, it then flows into the large, close (only *c.*14 km away), and stable water body of Lake Waahi.

1.12 Development of Spotlighting equipment

Various spotlight colour, and researcher approach techniques were analysed to determine the method that gave the greatest viewing ability, whilst causing the least disturbance to the fishes. Over 2 days, during daylight hours a 12 V spotlight fitted with an infra-red filter (transmission cut-off point at 790 nm) (Appendix (A), Figure 5.01/.) and remotely-controlled camera were suspended above pools in the Swanson Stream where a number of banded kokopu were known to reside. That following night behaviour of banded kokopu in the pool was viewed from approximately 25 m downstream via a portable 12 V black and white monitor connected to the infra-red sensitive camera (Plates 1.03/. & 1.04/.).

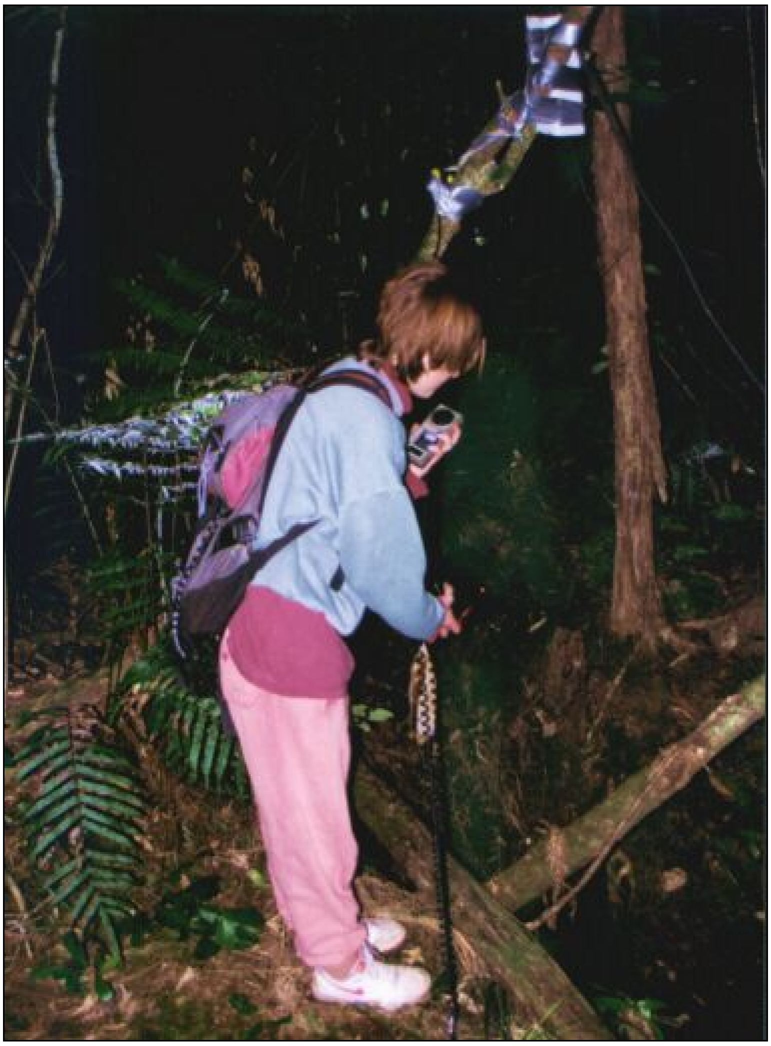

Plate 1.03/. Assistant approaching a pool containing banded kokopu over which an infra-red camera is mounted.

Plate 1.04/. Assistant observing the video monitor displaying the responses of the pool's banded kokopu to the various combinations of approach and spotlight characteristics.

Approaches to the stream were made over several runs after the fish had again resumed normal behaviour, in a varied combination of ways, and with different spotlighting methods and equipment. The effect of surveying variables tested on the fishes response included; approach noise, spotlight colour, and spotlight intensity.

1.13 Determination of bias calibration factor

Stream reaches of approximately 20 m in length were netted-off using 5 mm mesh stop-nets, attached on the sides to stream bank vegetation. Nets contained chains along their lower edge and were pushed into the sediment where possible. In the coarser sediments they had cobble piled up on the lower edge to again ensure no gaps remained. That night, a few hours after darkness and working from downstream up, all fish seen in the reach were counted using a rechargeable 12 V lead/acid gel battery powered Superlight brand model SL1705, 100 W halogen spotlight. The spotlight was also fitted with an optional analogue dimmer switch (set to a low intensity), a clip-on red filter (transmission cut-off at around 610 nm) (Appendix (A), Figure 5.01/.) and also had a variable focus capability (Fig. 1.05/.).

Plate 1.05/. The spotlighting equipment used in this study. From the top clockwise: 12 V lead/ acid gel rechargeable cells and pouch, variable focus 100 Watt spotlight, analogue dimmer control, infra-red filter, cherry-red filter, battery adaptor plugs.

For each fish seen, species and total length was estimated by eye and recorded; lengths being placed into 20 mm classes for banded kokopu (i.e. 0-19 mm, 20-39 mm,…etc. to a final class of 180 mm +). Total lengths of other species seen were estimated to the nearest 1 mm. In many cases banded kokopu were able to be approached so closely as to be able to hold a ruler directly over them to allow for more accurate length measurements (Fig. 1.01/.). The total time taken to count each reach by spotlight was also noted.

The following day, sample stream reaches were electric-fished with a National Institute of Water and Atmospheric Research Instruments (NIWA) Kainga brand model 300 backpack electric fishing machine. Although single-pass electric fishing has been shown to agree well with abundance estimates made by multiple passes (Richardson & Jowett, 1996), was fished to achieve at least a 50% reduction in banded kokopu caught (usually achieved with two passes). These extra passes increased estimate accuracy and enabled confidence intervals around the mean sample to be determined. Captured fish were placed in separate buckets of freshwater for each pass number, then anaesthetised with Benzocaine (ethyl amino-benzoate) prior to length measurement. The times taken to electric fish each reach was recorded, although data on the time taken to anaesthetise and measure the fish caught was not. Abundance estimates of banded kokopu populations made by 2- and 3-pass removal electric fishing in the reaches were calculated by formulae given by Armour *et al.* (1983), based on Zippin (1958).

Following Dollof *et al.* (1993), the calibration factor for spotlighting was determined as follows.

Firstly, the mean number of fish per 20 m reach section ($\bar{y}'$) was calculated;

$$\bar{y}' = \sum_{i=1}^{n'} {yi}/{n'}$$

where y_I is the number of fish estimate by removal electric fishing for each 20 m reach, and *n'* is the number of reach sections sampled. The mean number of fish counted by spotlight ($\bar{x}'$) was estimated by:

$$\bar{x}' = \sum_{i=1}^{n'} {xi}/{n'}$$

where x_I is the number of fish counted by spotlight in each 20 m reach sampled.

The calibration factor ($\hat{R}$) was then calculated for all sizes, and then both size classes by:

$$\hat{R} = \bar{y}'/\bar{x}'$$

1.2 Results

The ability to detect kokopu by spotlighting at night was negatively influenced by the presence of a broken water surface, depth, and also by water turbidity. Glare directly from reflection off the water's surface could be reduced by placing the spotlight on more of an acute angle to the surface plane of the water. This position, however, reduced light intensity through increased reflection from the surface, as was readily observed with a significant proportion of the beam consequently appearing in the forest canopy above the stream. The use of a cherry-red filter further reduced glare and backscatter due to suspended turbidity (especially fine clay suspensions) increasing both image resolution and beam penetration. Use of the red filter, however, also reduced the ability of the observer to detect unmoving kokopu via their colours and patterning, with only a gradient of red (c.f. grey-scale) subsequently being available during observations. This latter incapacity was largely easily circumvented with a simple identification of the regular and homogeneously textured shape of an inert fish on the stream benthos readily contrasting with the more irregular and varied texture of the substrate immediately beneath. Surface disturbance posed more of a problem with the possibility that estimates of numbers of fishes in reaches with a greater proportion of broken water surfaces may be being underestimated. The presence of koura in the streams during spotlighting also reduced efficiencies with their behaviour (even under very dim red light or even infrared light) of rapidly retreating backward via flapping of their telson in an effort to the evade the light, serving to disturb fine sediments and agitate any fishes that may be present nearby.

Combinations of spotlight characteristics and approach variables produced a range of responses in banded kokopu behaviour at night (Table 1.02/.) that appeared consistent between the pools and times in which the trial took place.

It was found that that the most ideal combination was for a red-filtered spotlight to be used on as dim an intensity setting as to still readily facilitate fish observations, but not so as intense as to agitate the banded kokopu. This was most effective when coupled with a quiet approach. Indeed a noisy approach had such an over-riding effect that most fishes were even already under cover and disturbed by the time the observer was near to the pool.

Table 1.02/. Summary of typical behaviour of banded kokopu to approach and spotlight type.

Approach noise	Spotlight colour	Spotlight intensity	Fishes typical behavioural response
Loud	White	High	Most fish already disturbed and hidden-
Loud	White	Low	before spotlighting begins,-
Loud	Red	High	with any remaining fish being-
Loud	Red	Low	highly agitated and mobile.
Quiet	White	High	Fish scatter rapidly and will not settle.
Quiet	White	Low	Fish move to bottom, often still unsettled.
Quiet	Red	High	Fish move to bottom, agitated; settle slowly.
Quiet	Red	Low	Fish move to bottom, settle quickly.

It was further observed that spotlighting efficiency was particularly less efficient for young-of-the-year, fishes (i.e. those which have spent less than a year in freshwater) than for fish greater than this size. Part of this was due to fishes of this age displaying a strong negative photo-taxic response upon the spotlight illuminating them in the stream, and part no doubt due simply to their significantly smaller size. It was also hypothesised that smaller fish may be exhibiting a different habitat preference to that of larger fish, possibly being more predominant in the shallower and faster waters such as riffles where they are less visible; a theory that has also been proposed by other researchers (Halstead, 1994; Dave West National Institute of Water and Atmospheric Research, pers. comm.). The bias calibration factor was therefore estimated *post hoc* following field data collection, for two size classes of fish in addition to the total sizes of fishes from the 20 mm class total length data that had already been collected. An independent bias calibration factor was thus calculated for the smaller (0+) fishes (total lengths exclusive to 60 mm prior to May 1997, and to 80 mm for results collected after this date) (Table 1.04/.). A single bias correction factor for the remaining (1+) fish (all fish with sizes above these total lengths) was calculated (Table 1.05/.), as was a calibration factor derived from the total size class data (Table 1.03/.).

Difficulty in electric fishing due to the size of the fishes, and the complex nature of the habitat in which kokopu were found in the day (especially severely under-cut banks, and tree roots and wood debris) reduced the efficiencies of the first passes. Second passes were typically not more than a 50% reduction of first passes, and this increased when captures were split into 1+ and 0+ sizes, compromising multiple pass population estimates.

Table 1.03/. Bias calibration data for all sizes of banded kokopu.

Regional location	Stream number / Reach number.	Visual count (xi)	Electric estimate (yi)	Area (m^2)	Density (fish/ m^2)
West Auckland	Swanson Stream 1/1	13	32.82	13.22	2.48
	Swanson Stream 1/2	6	8.17	12.80	0.64
	Swanson Stream 1/3	11	16.20	15.40	1.05
	Swanson Stream 1/4	15	12.80	25.60	0.50
	Swanson Stream 1/5	12	16.20	21.94	0.74
	Swanson Stream 1/6	3	2.18	10.73	0.20
	Stream Total	**60**	**88.36**	**99.24**	**0.89**
	Unnamed Stream 2/1	8	20.00	21.91	0.91
	Unnamed Stream 2/2	10	20.25	23.52	0.86
	Unnamed Stream 2/3	12	15.40	21.44	0.72
	Stream Total	**22**	**55.65**	**67.25**	**0.83**
Coromandel Peninsula	Unnamed Stream /1	11	30.15	12.08	2.50
	Unnamed Stream /2	17	29.04	9.17	3.17
	Stream Total	**28**	**59.19**	**20.19**	**2.93**
	Driving Creek /1	22	34.72	15.30	2.27
	Driving Creek /2	16	20.00	19.80	1.01
	Stream Total	**38**	**54.72**	**36.38**	**1.50**
	Sawmill Stream /1	7	23.14	16.10	3.32
	Sawmill Stream /2	48	53.48	25.40	0.91
	Sawmill Stream /3	16	25.79	22.21	1.16
	Sawmill Stream /4	14	21.13	18.01	1.17
	Stream Total	**85**	**123.54**	**81.42**	**1.52**
Hakarimata Range	Waipuna Stream /1	30	36.36	27.42	1.33
	Waipuna Stream /2	5	18.75	18.91	0.99
	Waipuna Stream /3	11	14.40	28.50	0.51
	Stream Total	**46**	**69.51**	**74.90**	**0.93**
$\sum_{i=1}^{n'} (xi)(yi) \big/ n'$		14.35	22.55	18.97	1.32
$\sum_{i=1}^{n'} i$	20 sites	287	450.96		
$\bar{y}'/\bar{x}'$		**1.57**	**1.57**		

Table 1.04/. Bias calibration data for 0+ fish only. n/a. = calibration not applicable as no fish seen.

Regional location	Stream number / Reach number.	Visual count (xi)	Electric estimate (yi)	Area (m^2)	Density (fish/ m^2)
West Auckland	Swanson Stream 1/1	7	27.00	13.22	2.04
	Swanson Stream 1/2	1	1.00	12.80	0.08
	Swanson Stream 1/3	6	4.50	15.40	0.29
	Swanson Stream 1/4	1	8.00	25.60	0.31
	Swanson Stream 1/5	2	4.50	21.94	0.21
	Swanson Stream 1/6	0	1.00	10.73	0.09
	Stream Total	**17**	**46.00**	**99.24**	**0.46**
	Unnamed Stream 2/1	4	12.25	21.91	0.56
	Unnamed Stream 2/2	3	8.00	23.52	0.34
	Unnamed Stream 2/3	5	14.43	21.44	0.67
	Stream Total	**12**	**34.68**	**67.25**	**0.52**
Coromandel Peninsula	Unnamed Stream /1	0	1.00	12.08	0.08
	Unnamed Stream /2	0	5.00	9.17	0.55
	Stream Total	**0**	**6.00**	**20.19**	**0.30**
	Driving Creek /1	5	12.25	15.30	0.80
	Driving Creek /2	4	9.00	19.80	0.45
	Stream Total	**9**	**21.25**	**36.38**	**0.58**
	Sawmill Stream /1	1	4.00	16.10	0.25
	Sawmill Stream /2	6	5.00	25.40	0.20
	Sawmill Stream /3	4	4.00	22.21	0.18
	Sawmill Stream /4	5	4.00	18.01	0.22
	Stream Total	**16**	**17.00**	**81.42**	**0.21**
Hakarimata Range	Waipuna Stream /1	11	32.00	27.42	1.17
	Waipuna Stream /2	3	14.29	18.91	0.76
	Waipuna Stream /3	6	7.20	28.50	0.25
	Stream Total	**20**	**53.49**	**74.90**	**0.71**
$\sum_{i=1}^{n'} (xi)(yi) \big/ n'$		**3.7**	**8.92**	18.97	0.48
$\sum_{i=1}^{n'} i$	20 sites	**74.0**	**178.42**		
$\bar{y}'/\bar{x}'$		**2.41**	**2.41**		

Table 1.05/. Bias calibration data for 1+ fish only.

Regional location	Stream number / Reach number.	Visual count (xi)	Electric estimate (yi)	Area (m^2)	Density (fish/ m^2)
West Auckland	Swanson Stream 1/1	6	12.50	13.22	0.95
	Swanson Stream 1/2	5	7.20	12.80	0.56
	Swanson Stream 1/3	5	12.00	15.40	0.78
	Swanson Stream 1/4	14	5.33	25.60	0.21
	Swanson Stream 1/5	10	12.00	21.94	0.55
	Swanson Stream 1/6	3	1.00	10.73	0.09
	Stream Total	**43**	**50.03**	**99.24**	**0.50**
	Unnamed Stream 2/1	4	9.00	21.91	0.41
	Unnamed Stream 2/2	7	12.50	23.52	0.53
	Unnamed Stream 2/3	7	1.00	21.44	0.05
	Stream Total	**18**	**22.50**	**67.25**	**0.33**
Coromandel Peninsula	Unnamed Stream /1	11	29.16	12.08	2.41
	Unnamed Stream /2	17	24.05	9.17	2.62
	Stream Total	**28**	**53.21**	**20.19**	**2.64**
	Driving Creek /1	17	23.14	15.30	1.51
	Driving Creek /2	12	16.00	19.80	0.81
	Stream Total	**29**	**39.14**	**36.38**	**1.08**
	Sawmill Stream /1	6	19.30	16.10	1.20
	Sawmill Stream /2	42	48.00	25.40	1.89
	Sawmill Stream /3	12	22.20	22.21	1.00
	Sawmill Stream /4	9	16.00	18.01	0.89
	Stream Total	**69**	**105.50**	**81.42**	**1.30**
Hakarimata Range	Waipuna Stream /1	19	16.00	27.42	0.58
	Waipuna Stream /2	2	8.30	18.91	0.44
	Waipuna Stream /3	5	7.20	28.50	0.25
	Stream Total	**26**	**31.50**	**74.90**	**0.42**
$\sum_{i=1}^{n'} (xi)(yi) \big/ n'$		**10.7**	**15.1**	18.97	0.89
$\sum_{i=1}^{n'} i$	20 sites	**213**	**301.97**		
$\bar{y}' / \bar{x}'$		**1.42**	**1.42**		

With the spotlight counts for all fish ages having to be multiplied by 1.57 to correct them to the electric fishing estimates for total size classes, we are therefore only seeing roughly 64% of the total numbers of fish present. It should be noted that this calibration value is sightly different from a previously cited value of 1.54 (i.e. 65% of actual fish present seen) (McCullough & Hicks, 1997) due to a now increased data set size. Following this same approach, mean proportions of only 0+ fishes seen at night are therefore 41% (1/ 2.41), and for 1+ fish alone 71 % (1/ 1.42). This alternative calibration factor for 0+ fishes, can be then be seen to be a notable improvement on estimates of populations where this size class is present.

Lines representing the value of the bias calibration factors for each age class were fitted through the origin as if no fish were present in the reach (i.e. 2+ pass removal electric fishing estimate = nil) then no fish would be expected to be seen (i.e. spotlight count = nil). The same logic was felt to warrant the use of the origin as the y-axis intercept with regressions of the different age classes also. Whilst similar, the regression method of bias correction yielded notably different results. Although untested for, given the low sample number it is very unlikely that any of these regressions would be found to be significantly different from any of the others. This is due to greatest variations coming from *within* streams (intra-stream variation) as a result of banded kokopu abundances closely mapping the high stream heterogeneity found, masking the lesser variation encountered *between* streams (inter-stream variation) (Tables 1.03/., 1.04/., 1.05/.).

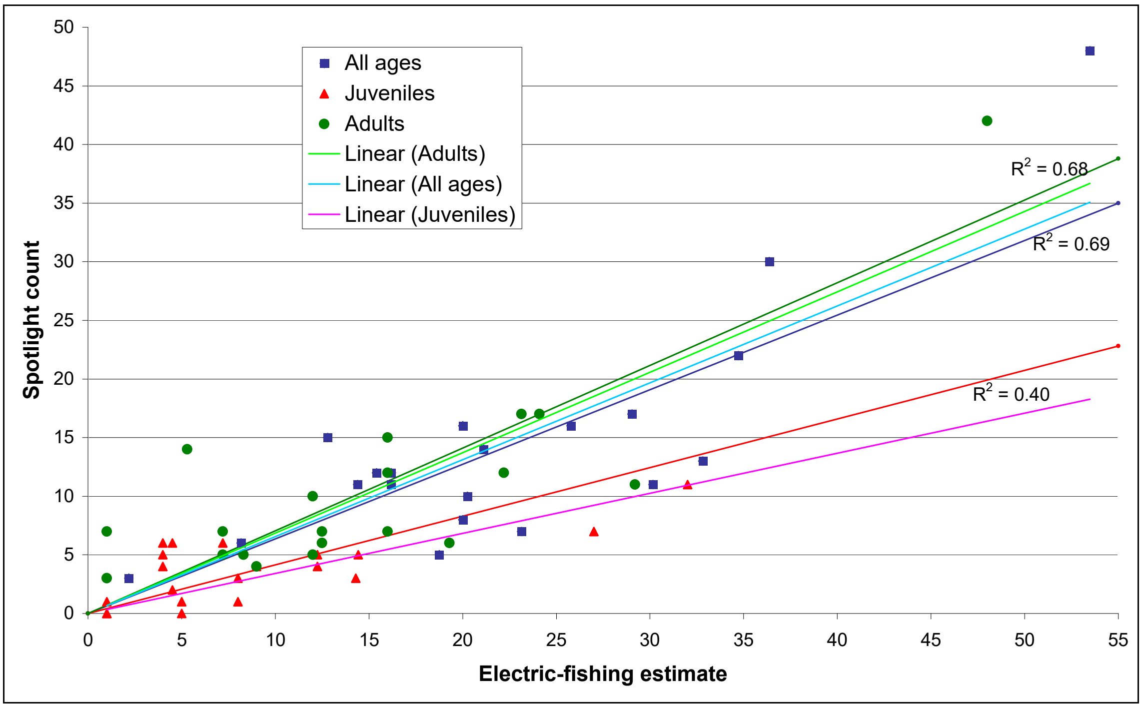

Figure 1.03/. Comparison of regression and ratio method for bias estimation. See text for details.

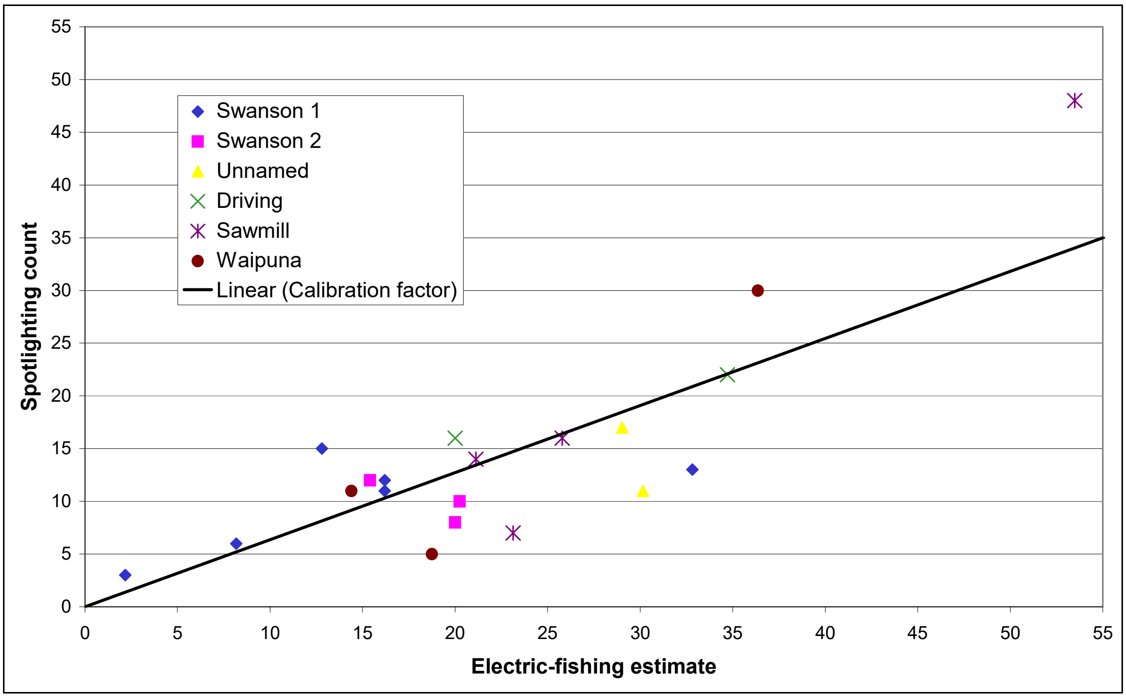

Figure 1.04/. Fit of sample stream scatter points around the mean calibration factor for all sizes.

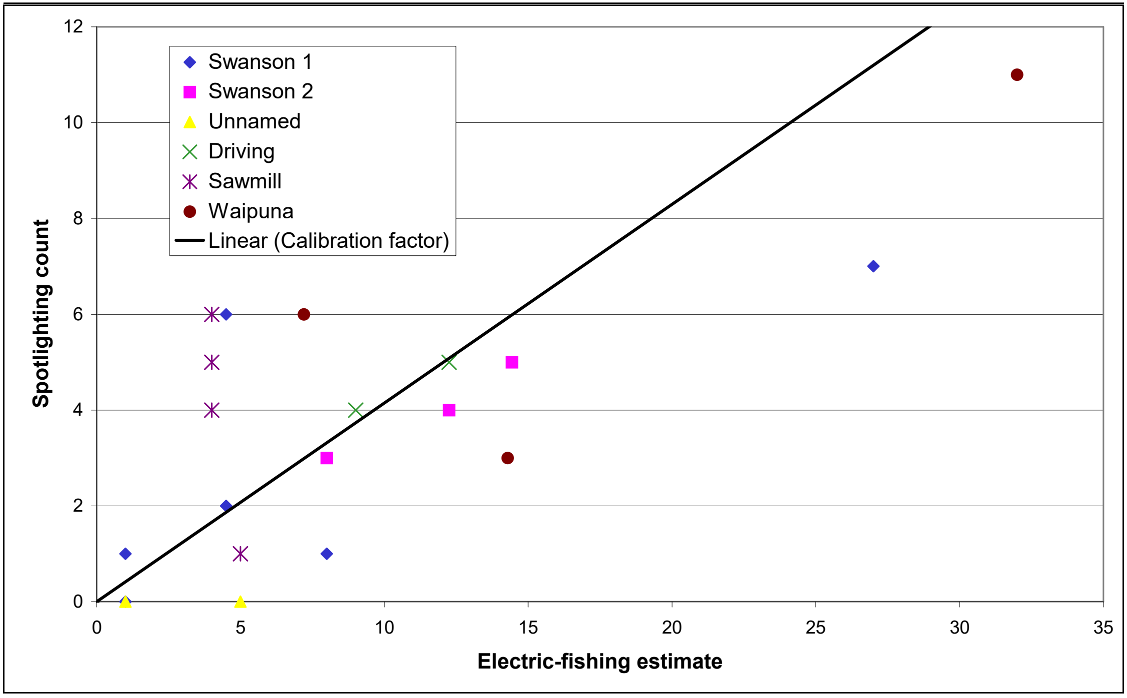

Figure 1.05/. Fit of sample stream scatter points around the mean calibration factor for 0+ fishes only.

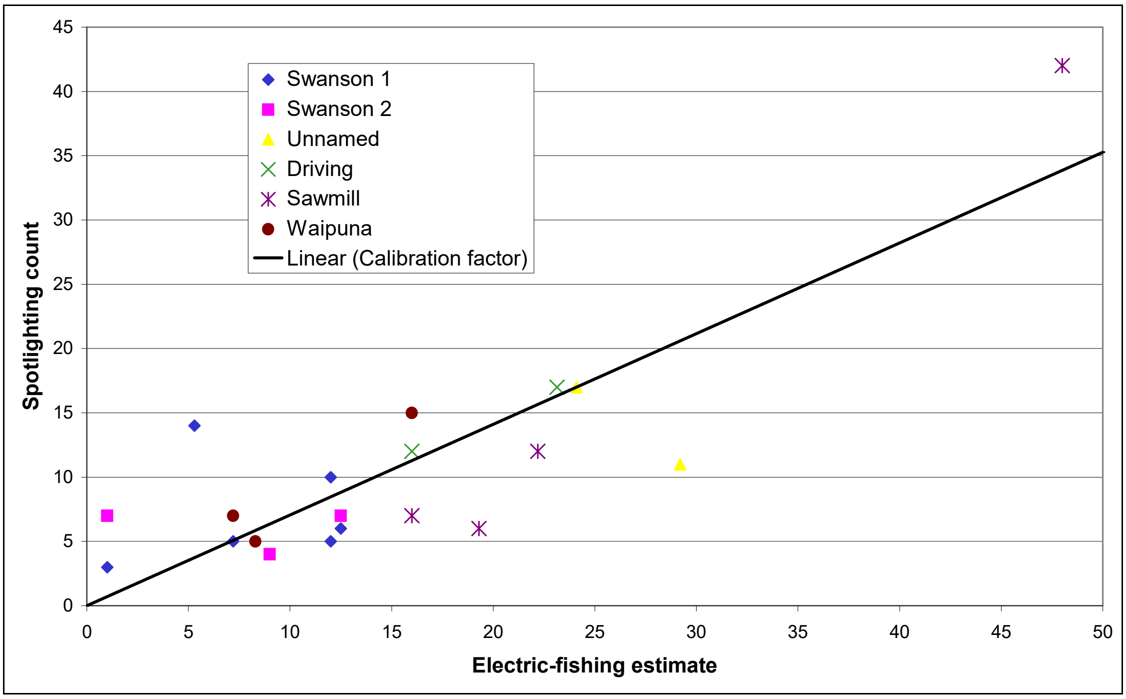

Figure 1.06/. Fit of sample stream scatter points around the mean calibration factor for 1+ fish only.

The mean total time/ m^2 taken to estimate each 20 m reach population by spotlight and electric fishing was significantly different (t-test, P <<0.01) with electric fishing counts for the same reach on average taking more than 6 times as long (Figure

1.07/.). This calculation did not include estimates of the time required to assemble stop-nets, or the time required to anaesthetise, measure, and return kokopu to their pools of origin.

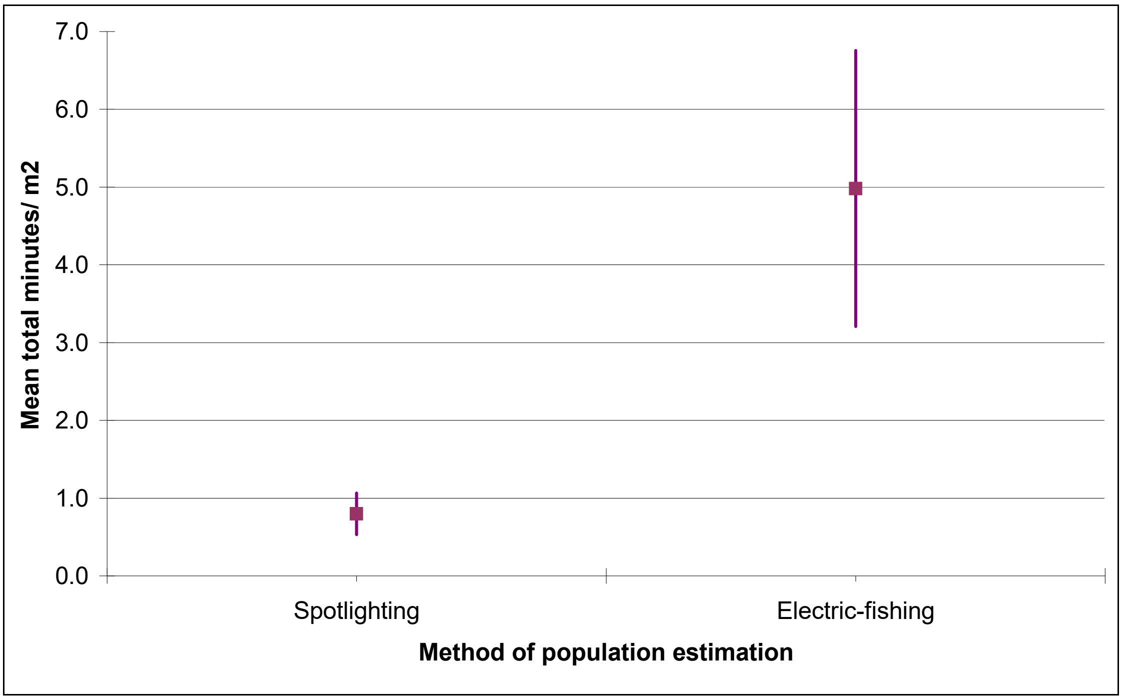

Figure 1.07/. Comparison of total mean surveying times (minutes taken/ m^2 stream area) between the methods of electric fishing and spotlighting. The point represent the arithmetic mean, whilst vertical lines represent 95% confidence intervals.

As seen from Table 1.03/., spotlighting efficiency was poor in the unnamed Coromandel stream, with fish being highly agitated and unapproachable, spending much of their time under cover. The sampling of this site is thought to have coincided with adults preparing to spawn, with there having been a large increase in the total numbers of banded kokopu seen in this reach over previous sampling 3 months earlier. This is possibly as a result of a downstream spawning migration (Ots & Eldon, 1975; Mitchell & Penlington, 1982; Mitchell, 1991; for a contrary view see Hopkins (1979b)). A very large proportion of this population also showed damage to their fins and dorsal surfaces, presumably from intraspecific territorial behaviour as large eels were not present (Dave West National Institute of Water and Atmospheric Research, pers. comm.) which have previously been incriminated for this damage (Hanchet & Chisnall 1991). Stomach samples also appeared to be much lighter during this time than previous, implying that feeding may also have been curtailed in this period (pers. obs.).

This hypothesis is further supported by most of the fish being found under cover and not actively foraging in open water as previously had been observed during preliminary spotlighting here 3 months earlier.

There was a great deal of variation between adjacent reaches in banded kokopu removal electric fishing estimates (Tables 1.03/., 1.04/., & 1.05/.). This variation appeared to be especially correlated with general stream morphology and instream cover, with reaches containing the greatest proportion of these two variables typically having the greatest banded kokopu abundances. Thus, reaches that were similar in cover and pool proportion tended to have a similar banded kokopu abundance.

Length and thus demographic data were most strongly inexact when calculating for the 21-60 mm, and 61-80 mm size classes. This error initially presented itself with an overestimation of the fishes present in the 61-80 mm class, and an underestimation of the smaller classes, and then with a gross change to the reverse as the author became more experienced (Fig. 1.08/.).

As seen in Figures 1.08/. the proportion of total fish seen under spotlight to the actual number caught is extremely similar between streams from very different geographical regions, for a given similar date (January to April 1998).

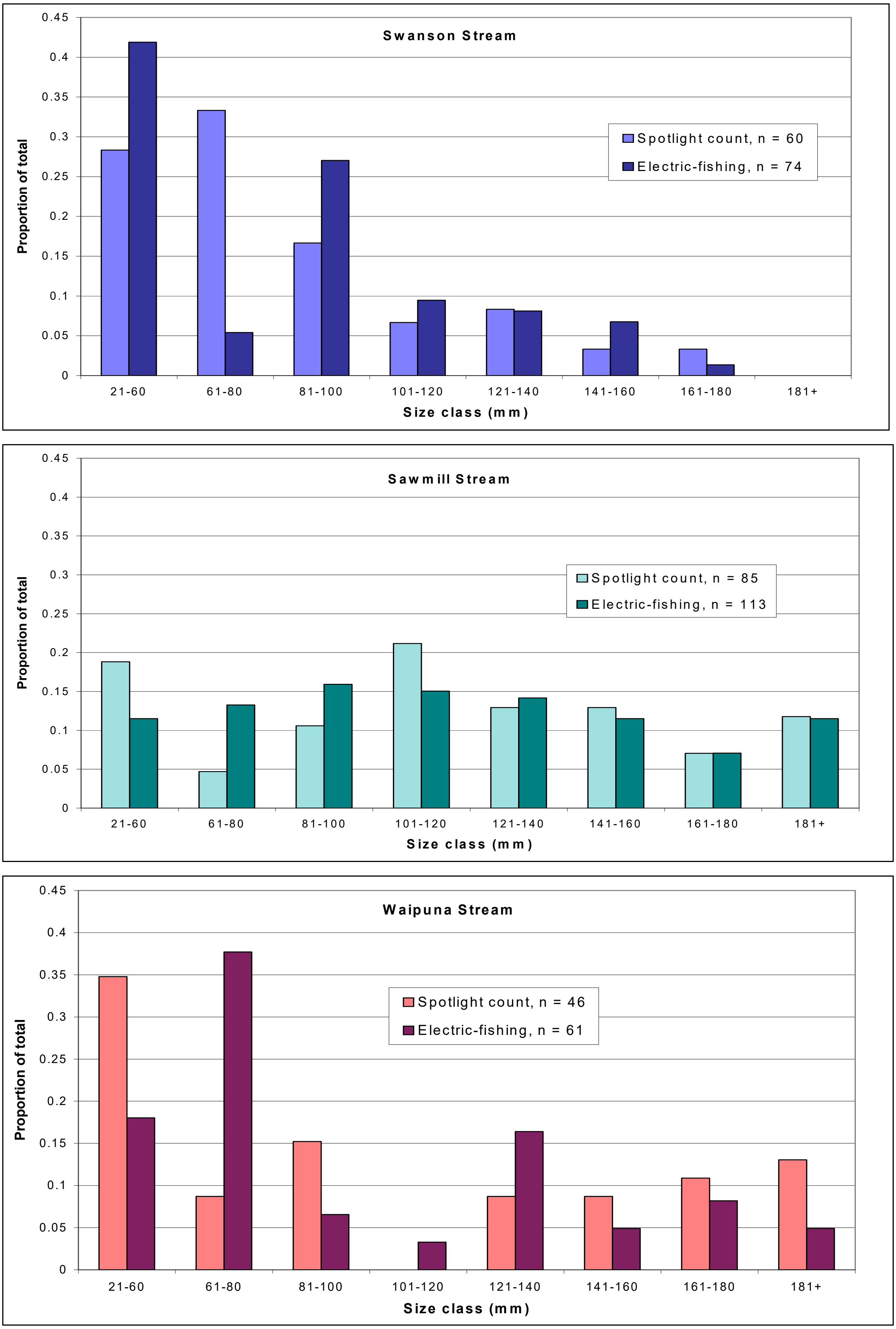

Figure 1.08/. Comparison of electric fishing counts with visual counts in Swanson, Sawmill, and Waipuna streams.

Other species noted and abundances estimated for

The streams sampled typically had a very poor fisheries diversity. The only other fish species seen whilst spotlighting included common bully (*Gobiomorphus cotidianus*) (only found in a single reach), and generally smaller individuals of longfinned eel (*Anguilla dieffenbachii*), and shortfinned eel (*Anguilla australis*). These last two species were also both frequently encountered again during electric fishing. Whilst initially counts were also recorded for these species both during spotlighting at night and during electric fishing the following day, logistics made recording this extra information impossible during the latter half of data collection. Preliminary analyses of the results from what data was obtained, implied that whilst spotlighted appeared to be efficient for common bullies (2 fish seen out of the 3 estimated to be present the next day by electric fishing), spotlighting efficiency was, however, poor for both species of eels. Although large individuals were often seen at night in a particular reach, these were frequently not recovered the following day. Smaller eels (less than 0.20 m long) presented the reverse situation; being only occasionally seen at night, but frequently caught during electric fishing in the day.

The results of this studies' total fish length/ frequency analysis from the electric fishing captures agree with those from previous studies both with wild animals in the field (Hopkins, 1979b), and with captive individuals in the laboratory (Eldon, 1969). Typical total maximum length attained in the next 12 months following migration from the sea to freshwater appeared to be about 60 mm. Although "shrinkage" of total length of the fishes occurs soon after they enter freshwater habitats (Hopkins, 1979b; Mitchell, 1982), given the assumption of fairly equal growth rates, growth appears to be fairly rapid thereafter as seen by the difference in plots of total fishes less the Waipuna Stream, and Waipuna Stream alone (Fig. 1.09/.).

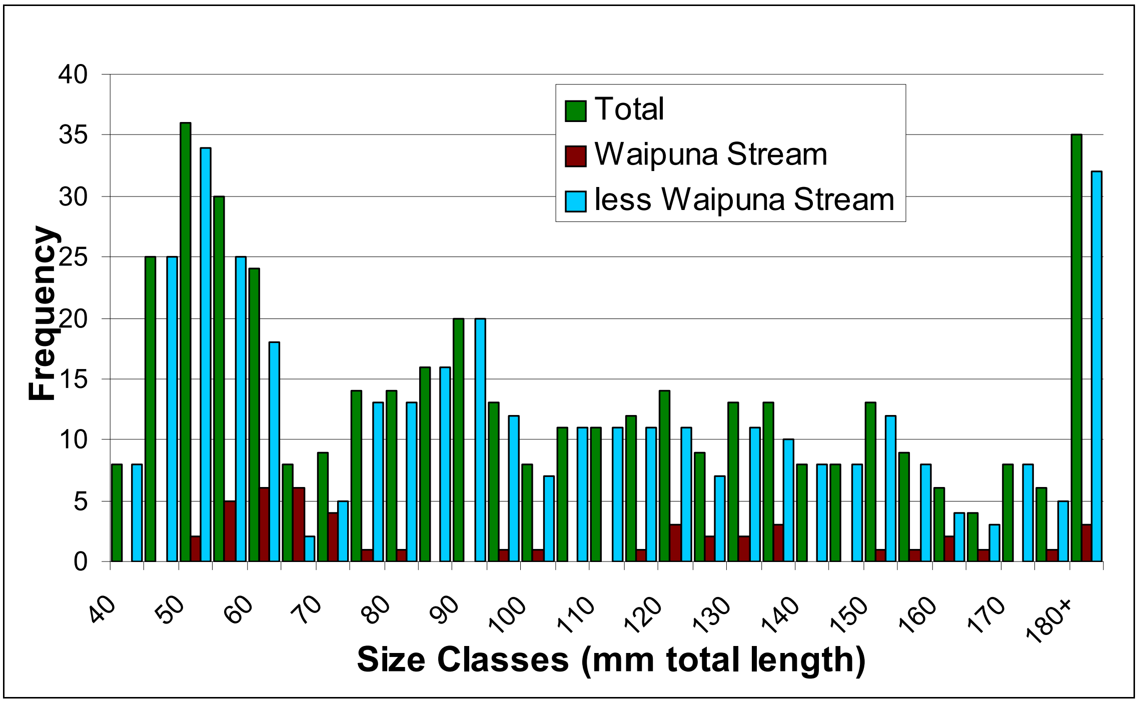

Figure 1.09/. Length/ frequency distribution of electric fishing results (total $N = 405$; Waipuna Stream $N = 47$; total less Waipuna Stream $N = 358$).

Via a demographic cross-sectional analysis, average growth rates were estimated from the mid-points of the highest frequencies in the length/ frequency graph (Figure 1.09/.). Estimates date of birth was followed after Hopkins (1979b) whom considered June to be the birth month of banded kokopu. This date is given by considering the gross majority of spawning to gave occurred in late April, and by then simply adding 5 weeks for the eggs to hatch, one month being required for egg development, and four months being spent at sea. Thus, 0+ fishes, as interpreted by the length measurements and date of sampling, would therefore be around 12 months old.

Table 1.06/. Estimated ages of banded kokopu from total electric fishing less later Waipuna Stream results (date around January 1997) ($N = 358$).

Approximate total length (mm)	Approximate age	
	total months	years old
55	6	0.5
93	18	1.5
118	30	2.5
135	42	3.5
153	54	4.5
173	66	5.5

1.3 Discussion and Conclusions

Spotlighting has shown to be a rapid and accurate tool for determining abundances of this nocturnal species. Given the three different calibration factors proposed, users of this method have a choice of taking a total abundance estimate simply from counting the number of banded kokopu seen in the 20 m reaches. It is, however, strongly noted that the sampling method is *equally* likely to detect all size classes evenly, thus the smaller size-class may appear to be under-represented or even absent when a single calibration factor is used (Hicks, 1985). Thus surveyors are recommended to further classify their data into 0+ fishes (less than 60 mm total length between September and April, or less than 80 mm between May and August), and 1+ fishes (all total lengths greater than those for 0+ fishes for these same dates).

Errors in estimation of stream fish populations appear to be more likely to be caused by spatial variations in distribution, than by the accuracy of individual point estimates of fish numbers (Hankin & Reeves 1988). It has been observed that there is a considerable variability in banded kokopu densities between reaches. Short sampling reaches will thus tend to underestimate the true variability in fish density distribution. With a rapid sampling technique such as calibrated spotlighting, uneven longitudinal distribution of fish is taken into account with an extensive count of predetermined reaches of the stream. Numbers of fish seen by spotlight counts are then corrected for comparison with the population estimates previous made by the removal electric fishing in 20 m sections i.e. the bias correction factor.

Thus, the bias correction factor $\hat{R}$ corrects for the bias of the spotlighting estimate to an estimate ($\bar{y}_{d,r}$) based on prior efficiency comparisons with electric fishing when multiplied by the number of fish seen by spotlight in a given section; i.e.:

$$\bar{y}_{d,r} = \hat{R}\,\bar{x}$$

Extrapolation for a total population estimate from sub-sampling of a reach length which is a multiple of 20 m ($\hat{Y}$) is simply by;

$$\hat{Y} = N\bar{y}_{d,r}$$

where N is the number of 20 m sections contained in the extrapolated reach.

Influences that may play a part in changing this spotlighting bias additionally are the reproductive state of the population in question, and also possibly water temperatures. Spotlight counting during the spawning months of April and May would appear to be significantly less efficient, with a consequent undetermined bias, than at other times of the year. Other than the observation of spotlighting being difficult with highly agitated fish when they were spawning, the dataset was of insufficient size to determine if any other seasonal effects may be present. Foraging activity which places kokopu in an exposed readily observable position may also be decreased in colder water temperatures, as is frequently noted with salmonids (Hayes & Baird, 1994). This is presumably due to metabolic reasons, such as reduced swimming performance at these temperatures.

Part of the difficulty with classification of the smaller classes (i.e. the 80 mm and under size classes) was thought to stem from the fact that their highly transparent tail makes up a large proportion of the body when they are at this length. It therefore follows that a miscalculation of where the tail actually ends (i.e. how long the tail actually is) will lead to a greater miscalculation of the fish at the smaller sizes, than when they are larger and the tail is both a smaller proportion of the total body length, and also displays a greater concentration of melanophores (Hopkins, 1979a). Thus if it is desirable to enhance accuracy of estimates of the demography of a given population spotlighting at night, it would be probably greatly enhanced by a prior “calibration” of the surveyor; maybe even by a simple prior exposure of the observer to fishes of a known length under the same light type proposed to be used.

The viability of spotlight counting for other species

Spotlighting at night is likely be a valuable technique also for other species in New Zealand. Spotlighting for common smelt (*Retropinna retropinna*) in small streams has also proved to be highly effective (unpublished data), albeit being difficult to distinguish this species from the similarly appearing inanga (*Galaxias maculatus*). No doubt the greatest restriction on the extrapolation of this methodology to other species will be the nature of the habitat in which the species is found; particularly the quantity of disturbed water surface present, or other observation confounding features such as excessive water depth.

A note on statistical power and the spotlighting method

Research and management questions, such as those involving these types of abundance data, are traditionally answered by the established method of statistical hypothesis testing, with a report typically being given of the α-value (the probability of making a type I error i.e. rejecting the null hypothesis (Ho) when it should not have been) (Zar, 1984). Little consideration, however, is ever given to the equally important concept of the β-value of the hypothesis test i.e. the probability of making a type II error, where Ho was not rejected when it should have been, often in the process drawing unjustified conclusions as a result. With the resultant conclusion typically being drawn that Ho is in fact true when it is merely not rejected, if there actually is an effect e.g. modification of riparian habitat *is* causing a decrease in the abundance of a species, then there is a very real danger that this effect will go unnoticed (Peterman, 1990). Furthermore, the false confidence given by this misinterpretation of the statistical analysis may encourage a blindness to any personal observations made that may suggest an effect is in fact occurring. Therefore with any hypothesis test, if Ho is not rejected, it is desirable to know the probability of having then possibly committed a Type II error.

The concept of statistical power is clearly of great significance to any person executing a hypothesis test. Statistical power is simply defined as 1 – β; where β is the probability of a type II error occurring. Thus an increased power value directly reflects the an increased probability of the analysis correctly rejecting Ho (Peterman, 1990) (Zar, 1984). A commonly recognised way of obtaining a higher β-value is simply through an increased sample size i.e. for a given α, larger samples will result in statistical tests with greater power (1-β). The only other (and less orthodox) recognised means to achieve a higher β-value is accomplished by departing from the arbitrary traditional α-value of 0.05, increasing it instead to a value of 0.1 or even higher (Zar, 1984). This is especially reasonable where (as is quite often the case) the cost of a type II error occurring is much greater than the cost of a type I error e.g. if a fisheries stocks were left to collapse as a result of over-harvesting not being correctly identified (Peterman, 1990). Therefore, with the likelihood of the spotlighting method being used as a management tool to provide data on population flux for hypothesis testing, it is important to realise the limitations of the resolving ability of this method to detect smaller population size changes.

Care should then be given to interpreting the results of a statistical analysis involving abundance data derived from spotlighting, that fails to reject the null hypothesis. It must be recognised that a failure to reject this hypothesis does not constitute a ordinance therefore to accept it. Thus practitioners of this spotlighting methodology must not assert in any way that no difference exists when a data analysis fails to reject an Ho.

Conclusions

Spotlighting may be seen to be both a desirable and a viable alternative to the current standard practice of electric fishing, when knowledge of the abundances of banded kokopu populations in streams are required. In addition to having many advantages (summarised in Table 1.07/.) for New Zealand's increasingly financially and personnel challenged local and national agencies with a freshwater resource management protection mandate, it also provides for a much larger sample to be attained, and with a consequent greater degree of realism and accuracy.

Table 1.07/. Synopsis of the comparison between electric fishing and spotlighting as methods determining population abundance estimates for banded kokopu in streams.

Method of Population Estimation	
Electric fishing	**Spotlighting**
Time consuming (mean = 5 minutes/ m^2)	Fast (mean = less than 1 minute/ m^2)
Expensive (about $7, 000 of equipment & c. $800 training/ person)	Effective, quality spotlight and rechargeable battery for under $500.
Labour intensive (minimum team of 2 required, heavy equipment)	Can be achieved with only a single person, light equipment assists travel over distance
Specialised training/ health requirements	Only elementary training required
Disturbance to habitat inevitable	Does not intrude into stream habitat
May damage and/ or stress fish	Less-invasive; minimal stress to fish

Whilst factors such as water velocity and water turbidity will negatively influence spotlighting efficiency (and therefore the accuracy of the bias correction) it must be remembered that these same factors will compromise electric fishing, in addition to others such as stream channel geology, water conductivity etc. (Anon, 1996a; Penczak *et al.*, 1997). For example a smooth water surface allows fish to be easily seen and recovered during electric fishing (Hayes & Baird, 1994).

Whilst valuable estimators for variance in spotlight determined are not yet available, the expected advent of these formulae in the near future will further increase the value of spotlighting as both a research and management tool for populations of banded kokopu.

Recommendations for the use of spotlighting in estimating abundances of banded kokopu populations.

1. Prior to surveying an orientation of the stream to be surveyed should be completed during daylight hours to familiarise the surveying team with the topography of the area and any obstacles or points of danger e.g. unstable or severely undercut banks. 20 m reaches to be surveyed should preferably also be measured and marked at this time.

2. Population estimates made by spotlighting during the months of late March into early May should be avoided due to spawning behaviours confounding results. Electric fishing should also be avoided during these months due to the sensitive nature of the fishes to stress at this time.

3. Surveying should be made in a direction from downstream upward to minimise disturbed and turbid water entering an unsurveyed section.

4. Standardised and simplified forms for recording purposes (such as those shown in Appendix (B), Tables 5.01/., 5.02/.) are desirable to avoid confusion and loss of data when working at night in sometimes difficult conditions.

5. Disturbance of the stream bank should be minimised, and stepping into the water also should be completely avoided where possible.

6. Spotlight light intensity should be high, and should also ideally pass through a red filter. This light should then be further adjustable to a maximum intensity allowable without disturbing the fishes, thus compensating for the differing disturbance thresholds of different populations. This intensity will be likely to vary both spatially between streams, and temporally between seasons.

7. The assistance of another person recording counts and taking other relevant notes (Plate 1.06/.) may cause a greater disturbance on the bank, but is likely to improve the estimate as the use of an extra person recording was a feature of the calibration methodology. Furthermore an extra person surveying is desirable from a safety aspect given the intrinsic dangers, compounded in a remote area, of travelling along a bush stream at night.

8. Whilst unquantified in this study, it is still clear that a higher value of α e.g. 0.10 will be accompanied by an increase in the statistical power of the population analysis. Given the ease and speed of the spotlighting method, consideration should also be given to collecting as larger sample as possible of the reach to be extrapolated to; which in addition to enhancing statistical power (smaller probability

of a Type II error), will also further serve to decrease the probability of a Type I error.

Plate 1.06/. Senior Biological Sciences technician Dudley Bell recording spotlight counts; the use of the headlight frees the hands for note taking.

CHAPTER 2.

Diel Activity and Instream Behavioural Observations

"The taxonomy, basic life-histories, and general distribution patterns of New Zealand's native fish fauna have been fairly well described. General features of "typical" habitats of most species also have been described, but much remains unknown about the behaviours of many species and the specifics of their habitat use."

Hayes (1996)

2.0 Introduction

Behavioural interactions of organisms with their environment are a frequently overlooked aspect of biological research; often appearing to be less applied than a more typical approach though directly empirical techniques. These methods may, however, provide simple and valuable insights into how a species utilises its physical and social resources. They may also further contribute to, and complement greatly with, information from the more archetypal ecological methodologies.

Whilst previous studies have delved into the feeding (Lyon & Main, 1988; Main, 1988; West, 1989), life history (Ots & Eldon, 1975; Hopkins, 1979a, 1979b; McDowall & Eldon, 1980; Mitchell & Penlington, 1982; Mitchell, 1991), prey detection physiology and observations (Halstead, 1994), or captive behaviour in aquaria (Eldon, 1969), these works have also disagreed with their description of the main type of diel activity displayed by this animal. Previous theories held range from banded kokopu being active all day in the gloomy conditions found in the dense undergrowth of native bush, and reverting only to a nocturnal existence when the habitat is more illuminated due to a loss/ change of cover type (Eldon, 1991) to observations that banded kokopu in streams were active all night (although peaks of activity were not noted in this latter study) Halstead (1994). This latter study, however, also failed to note what activity may exist during the day. Other works have even proposed banded kokopu being active during the day, and sleeping at night in the open pools to avoid predation.

Adults are sometimes heard in streams during the day as one approaches a stream. And juveniles, especially, are often seen fleeing ahead of an observer as one walks alongside the stream during the day, implying a more diurnal activity for this latter age class. Although these observations of these juveniles typically far outweigh those of the expectably more visible adults also likely to be present in the stream (pers. obs.), the easily disturbed nature of this species, however, greatly biases observations that are made with an observer present. Normal behaviours are also unlikely, in much the same manner, when the subjects under study are observed in unnatural circumstances such as aquaria. To circumvent this classic observer dilemma observations were thus made both without an observer present, and with wild animals *in situ* in a typical natural stream setting.

2.1 Methods

2.11 Data collection

Entire single day (24 hour) activity of banded kokopu was recorded twice in pools in the unnamed stream on the Coromandel Peninsula (Fig. 1.02/.) (18 August and 29 September), and once in a West Auckland (Swanson Stream) stream pool (10 October) in 1997. A 240 V infrared sensitive camera, mounted on a felled tree over a stream pool, was used to determine any peaks of activity for banded kokopu in pools. The same 12 V halogen spotlighting equipment fitted with a red filter used for the spotlight bias determination, was utilised again on a very low setting so as not to disturb the fish, but so as to be just visible to the variable iris Cosmicar brand camera set above (Plate 2.01/.). This same arrangement was run throughout the recording cycle, during daylight hours and at night.

Plate 2.01/. The low-light sensitive video camera mounted on a felled tree over the Swanson Stream (West Auckland). Note the spotlight mounted along a different plane to the camera to reduce glare off the water's surface.

Banded kokopu behaviour observed with the red light spotlight appeared not to be different to their behaviour as that found under viewing with infra-red light (filter cut-off point 790 nm) (Appendix (A), Figure 5.01/.) which is believed to be visible to most fishes (Lythgoe, 1979). Use of the red filter permitted a greater proportion of the emitted spectrum through the filter, allowing the spotlight to run on a lower power level requirement. This substantially extended the life of the generator's fuel tank before requiring refilling. This was a significant issue, with the generator further being fitted with a ancillary 3 l fuel tank to extend its operating time from 3 hours to 12 hours without refuelling. The power and signal cable from the camera ran back 15 m to a monitor and a Hitachi brand VTL2000E time lapse VHS recorder/ player, which were then housed in a waterproof box. The 24 hour activity sample was then recorded onto a normal VHS E180 minute tape (Plate 2.02/.).

Plate 2.02/. Ian Jowett (National Institute of Water and Atmospheric Research, Hamilton) examining the time-lapse video recording apparatus in its waterproof container (Unnamed Stream, Coromandel).

The entire system (spotlight source, camera, black and white monitor, and recorder) were all powered by a Honda brand model EX350 2-stroke generator equipped with a 3 litre fuel tank, with a entire 24 hour (entire day) session being recorded onto a single 180 minute VHS tape, and refuelling only required once every 12 hours. Sites examined were a pool in Swanson Stream, and the unnamed stream on the east side of the Coromandel golf course. Sites were specifically chosen for the low amount of daylight incident on their surface waters (causing glare) and their high abundances of banded kokopu. Both sites had a water velocity though them of around 0.05 to 0.10 m/s, and a still backwater to one side of the field of view. Additional, complementary, behavioural observations were also made of captive banded kokopu (120 mm) kept in a 70 l aquarium with darkened back and sides, and a water velocity of *c.*0.20 m/s across the front glass panel.

2.12 Data analysis

180 minute video recordings were played on a standard VHS recorder with the number of banded kokopu seen to be present in open water (i.e. foraging) in the picture being recorded. The area of the picture where banded kokopu were counted was limited to the minimum area that had either been visible at night (due to the spotlight not illuminating the full viewable picture), or what had been visible during the day (especially due to glare permitted by incomplete canopy closure). This reduced counting area was marked on the TV screen with lipstick to maintain a constant area of view, and thus avoid a bias when assessing activity for a different time of day with differing viewable picture areas.

2.2 Results

Of the three video transcripts made, none were complete for a 24 hour period. The Coromandel 1 stream transcript was incomplete due to the generator only having a very limited fuel supply at this point, and therefore requiring frequent refuelling. The other recordings were both affected by rainfall causing the stream to become turbid. Further recording attempts were discontinued due to rain later in the exercise. The fragmented results from each recording, and their averaged activity were then combined together to display a general mean activity and the range of the source data (Fig. 2.02/.). This was complete for a 24 hour period, with gaps of data for one time of one transcript being filled by another.

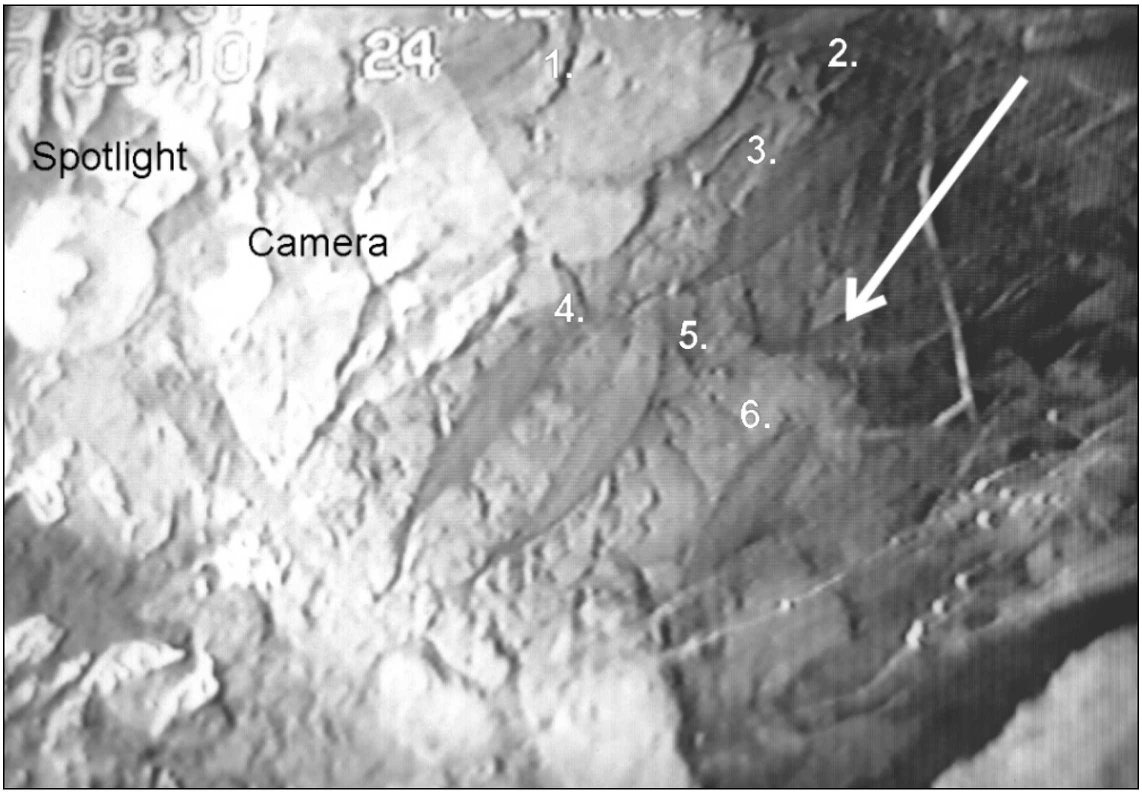

Plate 2.03/. Video transcript of early evening (5:02 p.m. and still daylight) activity in the unnamed stream, Coromandel. Six fish are visible, all oriented and swimming slowly to maintain direction in the flow of the stream. The reflections of the spotlight and camera are visible in the upper left corners of the picture.

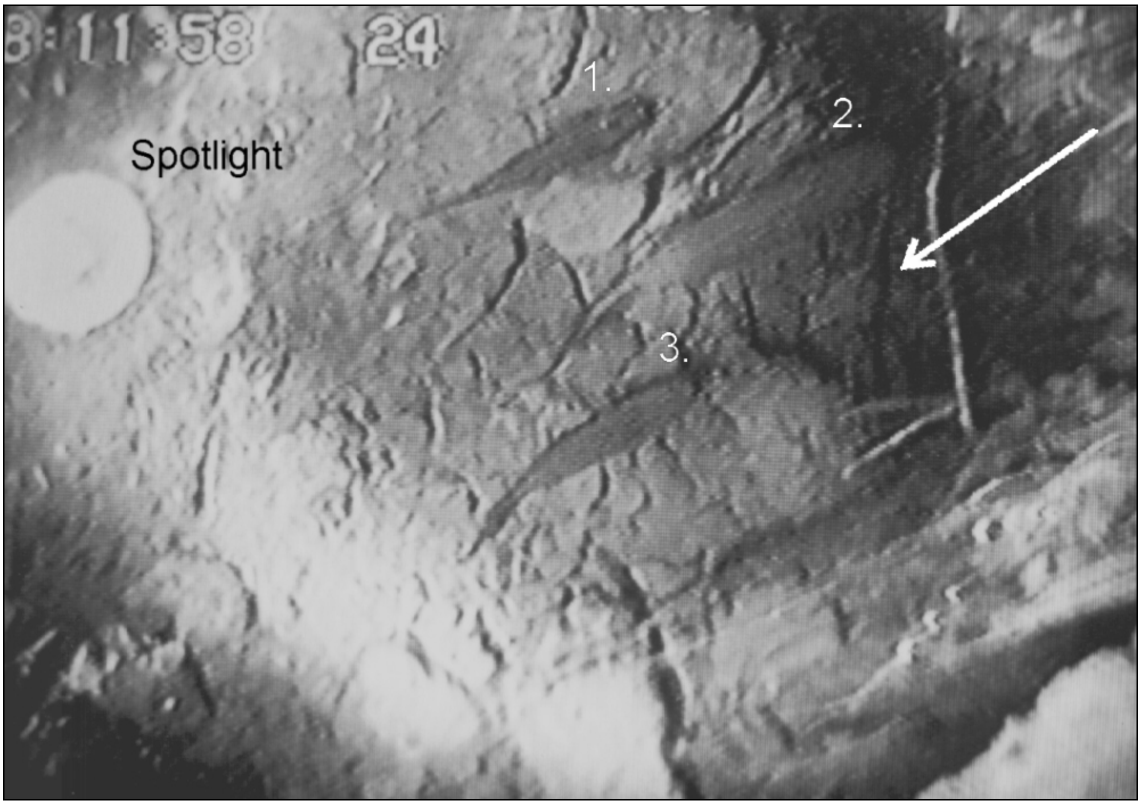

Plate 2.04/. Video transcript of the same site taken 1 hour, 10 minutes later than Plate 2.03/. with the spotlight now more visible.

Activity

Greatest activity recorded from all streams appeared to be around the hours early to late afternoon, and early the next morning (Fig. 2.02/.). Although the data set is small, the major disagreement between separate transcripts activity recordings appeared to be mainly in temporal shifts of peak activity to a few hours either earlier or later in the day of another recording's. Whilst in general evening activity recordings between the three different dates and two different locations agreed with each other, there was, however, insufficient data to determine agreement for early morning activity.

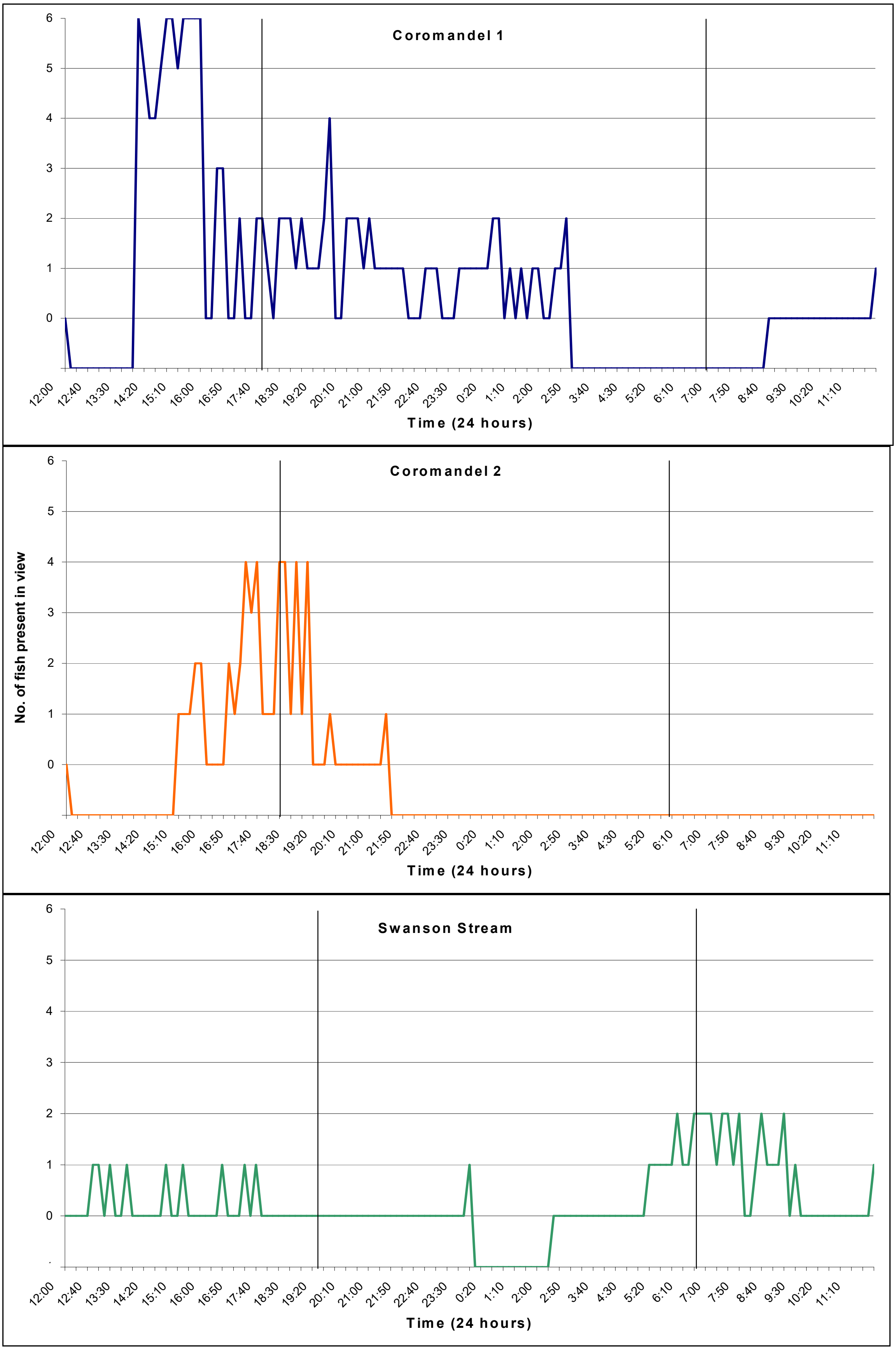

Figure 2.01/. Diel activity of banded kokopu in streams for each video transcript. Black vertical lines indicate sunrise and sunset for that series. The series line dropping below 0 indicates that no data was available for that particular series at that time.

Major peaks of mean activity (Fig.'s 2.01/., 2.02/.) appeared to be around early morning (6:00 to 8:30 a.m.) and early afternoon to late evening (2:00 to 7:30 p.m.). Notable differences were observed between video transcripts, especially for the two transcripts taken at the same Coromandel location. For these two same site recordings, there appeared to be an apparent shift of peak activity from a beginning at around 2:00 p.m. to a later time of 3:30 p.m. in the transcript taken from a date 8 weeks later. By far the majority of the diurnal activity was due to 0+ fishes, usually maintaining position and swimming at the head of the pool, with all the mid-afternoon activity seen by in the Swanson Stream recording being due to a single 0+ fish. Although found under cover when the first recording was being set up (around 2:00 p.m.), when a stick was inserted into cover at the side of the prospective pool to be recorded (to ascertain the presence of banded kokopu there), two adults responded immediately by attacking it before realisation of the true nature of the "prey item", and their consequent retreat.

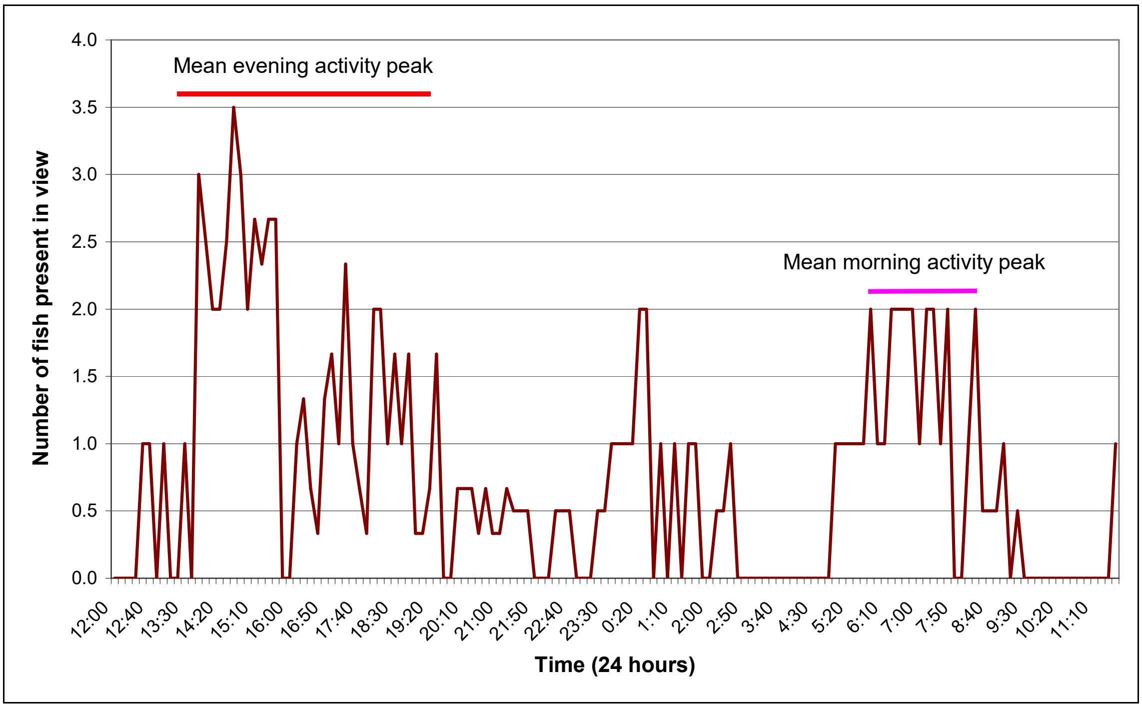

Figure 2.02/. Mean diel activity of banded kokopu in streams for all video transcripts.

Observed behaviour of banded kokopu

Whilst Halstead (1994) had found that banded kokopu remained separate from each other with no evidence of shoaling, the video transcripts showed that they were often found maintaining position at the heads of pools in groups, each individual slightly offset and either forward or back of its neighbour (Plates 2.04/., 2.05/.). This differences in observed behaviours may be due to there being a higher density and size of fishes present in the Coromandel stream than the West Auckland stream of Halstead (1994). There was also a stronger current present through the narrower and more definite pool in which the Coromandel observations took place (pers. obs.) potentially causing a further bias toward active swimming. This foremost position in the pool was often held by a large individual.

Cobbles and bank outcrops were also often favoured habitat items with the fish placing their focal point immediately below these instream obstacles to flow. Backwaters to the main flow at the sides of the pool were also favoured, with their lower water velocities. Good use was made of shallow areas to the side of pools where kokopu often “beached” themselves against the current, facing upstream often with their back partially out of the water. One fish was even seen to have deliberately propped its tail against a small cobble in shallow water at the head of the pool, and thus braced and oriented itself into the direction of the current with its head near the water’s surface.

Banded kokopu appear to be transient swimmers (Webb & Blake 1985) capable of good bursts of speed when engaging prey and avoiding predators with the posterior positioning of their dorsal and anal fins increasing the force for a caudal tail-fin movement. The dorsal and anal fin appears to be collapsed against the body during cruising, reducing drag, and being erected mainly during accelerations and turns. Pectoral fins were also held against their sides during cruise swimming in the current. Maintenance of position in weak currents is often by a flexible wriggling of the caudal fin in figure-8 type shape (Main 1988; pers. obs.) when viewed from the posterior.

Almost all activities (reasonably presumed to be foraging) occurred at the surface of the water, as was easily noted in the recordings by the dorsal or caudal fins occasionally breaking the meniscus. As was noted with Halstead’s observations (1994), downstream drifting prey and vegetation debris on the surface of the water appeared to be the primary interest. These were typically detected at a distance of between one and two body lengths (*c.*15-30 cm) away from the focal point of the fish, rapidly approached and acquired, and then accepted or rejected presumably as a function of their taste.

Heavy rain prevented activities being recorded during some of the transcripts due to surface disturbance and fine sediment mobilisation. Rain was also seen to disturb fishes foraging, as large raindrops impacting on the surface of the water were immediately followed by kokopu fleeing to the nearest source of cover; typically root wads and the undercut banks of their pools. Small sticks and vegetal debris impacting on the water's surface, either as a consequence of wind or of rain also elicited this prompt evasive reaction. Feeding, however, was seen to continue during light rain which, whilst compromised, still allowed for sub-surface visibility.

Considerable use appeared to be made of the undercut banks and their combined root-wads of trees, with kokopu returning to under this cover, found in a still part of the pool, between their bouts of feeding activity. Burrowing into sandy substrates under rocks has also been noted with captive banded kokopu in aquaria, usually with a swimming action of the caudal fin removing debris originally excavated by the pectoral fins. kokopu are often then seen with their head only protruding from this newly created cave, rapidly emerging to seize food items, or to feed in the current.

Little intraspecific aggression was observed between individuals of a similar size in recordings, and little aggression has also been noted toward other species kept in aquaria. What aggressive behaviour that has occurred has usually been in response to attacks from other species present e.g. crans bullies (*Gobiomorphus basalis*) defending their territories. However, whilst large fish occasionally chased smaller fish from the heads of pools, more notable aggression was also noted at one point during recording (18 August, Coromandel 1 record). Although some months after spawning, this took the form of what it no doubt responsible for many of the injuries noted in this stream in April; namely bitten caudal and pectoral fins, especially on large adult females. At around 5:00 p.m. a large disturbance was noted immediately upstream of the observation site, betraying itself by a plume of fine silt being swept down in the current and first thought to be due to a foraging eel (*Anguilla* spp.). Soon after, smaller actively foraging kokopu present in the picture fled to the side of the picture and bank cover as a pair of large adult female tumbled past; one individual locked onto the right pectoral fin of the other.

Additional observations of koura

In the single recording made of them, koura (freshwater crayfish) *Paranephrops planifrons*, appeared to be truly nocturnal, being active only during the hours of darkness. Foraging activity was first observed around 10:00 p.m., with a final observation at 7:00 a.m. soon after the sun rose. While koura were present in the Swanson Stream video transcript, however, no koura were seen in the Coromandel stream recordings. Koura were seen to interact with each other, apparently demonstrating a territoriality with a single individual aggressively dominating a territory against other foraging koura. As with Halstead (1994), foraging crayfish were also seen to exhibit aggressive behaviour toward individuals of banded kokopu, with chelae being raised as a fish approached. This aggression, however, appears to be reciprocated with a middle-sized kokopu chasing a middle-sized koura during the early hours of the morning in Swanson Stream.

2.3 Discussion and Conclusions

The results of this study would imply that, as the original hypothesis had predicted, there are two peaks of diel activity seen with banded kokopu in streams. Whilst this activity occurs mainly outside of the times of the most intense and greatest degree of incident light (i.e. when the sun is lower and the streams are more likely to by shaded by their riparians) there also appears to be some activity in the middle of the day. Other authors have also noted from personal and anecdotal experience that banded kokopu tend to be primarily nocturnal except in dimly lit forest (McDowall, 1990; Parkinson & Cox, 1990) where individuals may also be found to be active in the day.

The aforementioned absence of fishes from the pool in which the camera was being installed during the day, yet their consequent rapid emergence from adjacent cover when the presence of prey was suspected support these observations that these fish are active and feeding during the day when in dimly lit conditions; yet readily flee when disturbed, such that their habitat consequently appears vacant. Although approach noise had been noted as a meaningful warning stimulus (Chapter 1) it would appear that their visual sense is, however, of primary use during times of the day in which sufficient light is available. Approaches at night therefore are more effective given the lack of specialised night vision on the part of the fish (Halstead, 1994). Furthermore, during this study streams with a known resident banded kokopu population could be approached successfully as long as approach disturbance was kept to a minimum, and most importantly; that the observer was hunched over. This technique enabled the height of the stream bank, and refraction of the observer image across the air/ water boundary, to effectively hide the observer allowing them to remain invisible and yet able to observe undisturbed diurnal behaviour. During this time banded kokopu would readily detect and attack stones or other items flicked across the surface of the water (followed by consequent rejection of the item), or even sticks dipped from the edges of the pool (Fig. 2.05/.). Populations in stream with frequent human disturbance appeared to be habituated, as they were the most approachable.

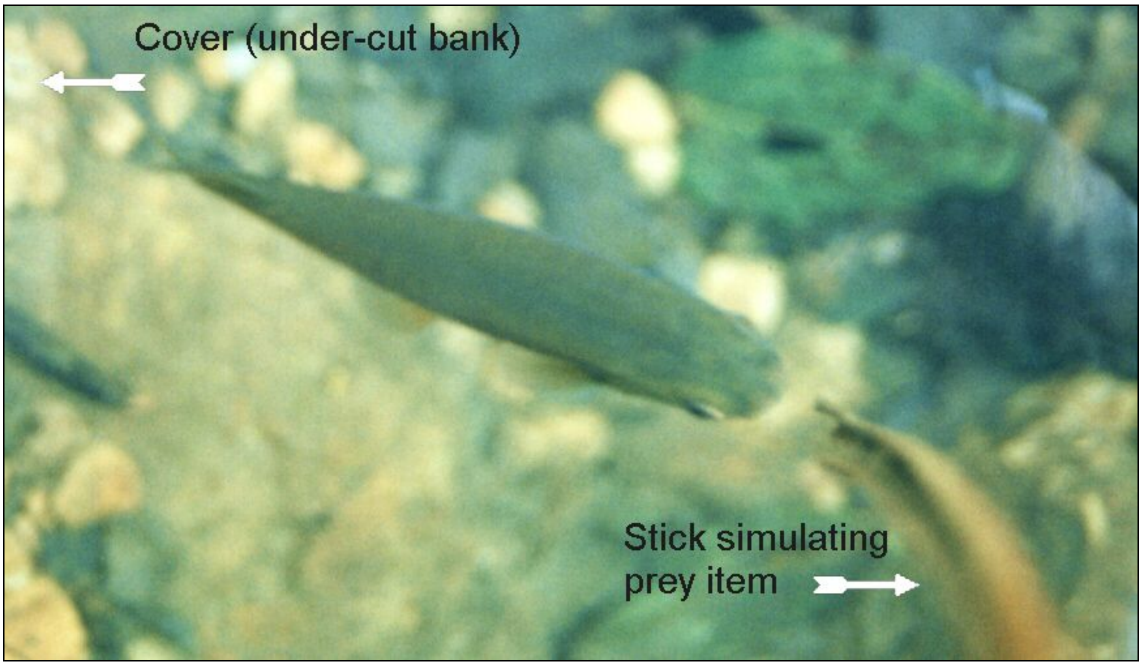

Plate 2.05/. Banded kokopu rising from bank-side cover approach in the afternoon to attack a small stick placed into the water following an unobtrusive. Buffalo Creek, Coromandel.

The apparent shift of peak activity in the first Coromandel recording (Coromandel 1, Fig. 2.01/.) to a later time in the later transcript (Coromandel 2) may be due in part to the confounding effects of the marked difference in dates and consequence sunrise, sunset and total daylight hours present during these different recordings. Whilst sunrise was over one hour earlier in the first (18/08/97) recording, it set almost half an hour later than this former records clock on the date of the second recording (29/09/97) (Lamont, 1997). Evening peaks of activity were yet similar with each other.

The possibility of 0+ fish activity occurring during the day, as seen with both study sites, may be a form of temporal segregation of the large and potentially piscivorous adults occupying habitats displaying preferred velocities and depths and the vulnerable smaller 0+ fishes. Juveniles of the mudfishes (*Neochanna* spp.) have also been noted to be active and feeding during the day before metamorphosing into nocturnal feeding adults (Eldon, 1991). Whilst classification into the classes of 1+ and 0+ fish used earlier (Chapter 1) as defined by total length was not possible in this study, plotting of the activities of separate age groups may yield a clearer answer to if this strategy is present or not.

It is a reasonable assumption that the two main peaks of activities observed would be oriented toward optimising prey acquisition available during the day. This leads us to several different functions which may be the basis of the relatively increased feeding behaviour during noted during these times of the day. Although Edwards & Huryn (1995) found that their was no significant difference between night and day for the total biomass of drifting invertebrates of terrestrial origin stomach sampling has found that total stomach contents are significantly heavier in the morning than contents taken at late evening (Halstead, 1994), implying that the majority of feeding occurs overnight.

Innumerable international studies have shown that sub-surface drift increases during the night, especially during the period just after sunset and just before sunrise (Brittain & Eikeland 1988). This nocturnal peak is supported by New Zealand work which found mayflies of the genus *Deleatidium* to comprise the greatest proportion of drifting invertebrates and furthermore that larger insects in general drifted predominantly at night also; presumably a behaviour oriented at avoidance of visually feeding fish predators (Watson 1971; Sagar & Glova 1992).

The distinctive “head” of the pool which the features of fast riffle water meeting the deeper reservoir of the pool formed, would have created a desirable position in which to intercept any downstream drifting invertebrates of terrestrial or aquatic origin as the water velocity slowed upon entrance to the pool Salmonids, particularly, have been noted to select and defend territories which are best suited for the interception of drift; the size and location of this territory being determined by the drift density and patterns of drift in the water currents. The use of this position whilst drift-feeding is therefore typical of salmonids such as brown trout (Fausch 1983; Hayes & Jowett 1994), and also by the closely related koaro (*Galaxias brevipinnis*) (Hayes 1996). This latter observation has also been noted to occur in the evening, as had been found to be a peak of activity for banded kokopu in this study.

Although drift-feeding was a particularly noted foraging behaviour in the recordings of this study, no doubt this is largely a consequence of the positioning of the camera at the head of pools. Non-drift feeding foraging behaviours are likely to be more important in habitats of the stream displaying slower water velocities, or in more streams displaying more lentic characteristics altogether. In these environs surface feeding on allochthonous terrestrial sourced invertebrates is likely to form the dominant behaviour type, with quite possibly a different activity periodicity thus reflecting the availabilities of these different prey items. It is also possible that kokopu show an emphasis on drift-feeding as recorded by increased activity in pool headwaters during these times of the day when drift is high, with foraging efforts changing to surface feeding in slower waters as the efficiency of drift-feeding becomes reduced following a decrease in the biomass of drifting prey. Changing behaviour from a visual prey-detection strategy to that of a prey-disturbance mediated detection is likely to be a common behavioural feature as encroaching darkness inhibits prey location by visual means.

Banded kokopu display a morphological feature known as a “lateral line” along the sides of their bodies which aids detection of surface prey, being especially sensitive to those struggling in the water’s meniscus (Halstead, 1994). Recognised as primarily a surface feeder, this lateral line (also commonly found in many other fishes) is probably mainly responsible for these detections of struggling allochthonous invertebrates creating a disturbance at the water surface, and is further assisted by the presence of cephalic laterosensory pores around the head (Bleckman *et al*., 1989; Halstead, 1994; McDowall, 1997a). In addition to this line and head pores, it has also been surmised that the an accessory lateral line along the dorsal additionally assists the better-developed lateral line found along the lateral edge of the animal (McDowall, 1997a).

Beetles (Order Coleoptera) have been found to form a significant component of the diet of banded kokopu (Main & Lyon, 1988; West, 1989; Eldon, 1992; Halstead, 1994; Hicks, 1997) as has also been found with the closely related koaro (*Galaxias brevipinnis*) (Main & Winterbourn, 1987). These contributions are significant albeit gravimetrically rather than numerically as beetles are substantially heavier than many other more commonly recorded prey items e.g. ants (Order Hymenoptera, Family Formicidae). Some families of beetles in the diet of banded kokopu (notably the Family Carabidae (Main & Winterbourn, 1987)) are primarily active during evening and early into the hours of darkness (John Early, Auckland Museum, pers. comm.). The same is true of weta (order Orthoptera, family Stenopelmatidae). Whilst not featuring abundant numerically in the diet of banded kokopu, these native insects also make up a significant gravimetric proportion of the diet (Halstead, 1994) due to their large size. These nocturnal animals abound around streams in native bush (Miller, 1984) and when disturbed readily flee across the water's surface striking the surface film abruptly with their hind legs (pers. obs.). Furthermore adults of many aquatic macroinvertebrates e.g. the caddisflies and mayflies (orders Trichoptera and Ephemeroptera respectively) typically also show increased mobilities during the evening (Kevin Collier, National Institute of Water and Atmospheric Research, pers. comm.) and are therefore increasingly likely to be found struggling in the surface drift through misadventure upstream. Drift feeding may therefore also be related to the increased presence of these terrestrial animals in the surface drift, in much the same way as it likely is to drift of aquatic invertebrates. Moreover the apparent greater length of time spend actively foraging at night relative to in the morning, may represent a greater proportion of prey availability found at this time, simply due to these early nocturnal invertebrate behaviours already noted.

Further advantages to surface feeding during these times of day are that they represent a transition from a higher land temperature regime encountered during the day (with consequential "on-shore" winds resulting), to a higher sea temperature at night (with subsequent "off-shore" winds). During this period of transition, sea and land temperatures are in a approximate equilibrium with each other and winds die to the well known evening and morning still periods. With wind across the surface of the water creating a “white-noise” or irregular background disturbance, feeding at the surface during these quieter times when less wind was present would facilitate readier prey detection and capture. Foraging behaviour, observed to continue even during light rain may, however, be facilitated by a neural filter which habituates to a repetitive predictable disturbance such is also found with banded kokopu living in pools close to small waterfalls (John Montgomery, University of Auckland, pers. comm.). The random nature of heavier or intermittent rain appeared to disturb the resident banded kokopu whom were then observed to scatter to cover during these occasions.

Crustaceans other than koura such as the ubiquitous and frequently abundant freshwater terrestrial and aquatic amphipods (Order Amphipoda) such as *Paracalliope* spp. also appear to play a significant part in their diet (West *et al.* in prep.) in streams which typically have a low benthic invertebrate faunal biomass (pers. obs.). With banded kokopu often seen to be singly feeding from the surface of the benthos, it is likely that these may be the primary prey being sought; although chironomids (Diptera: Chironomidae) have also been noted in dietary samples (West *et al.* in prep.). Benthic caseless caddisfly larvae (Order Trichoptera) e.g. *Polyplectropus* and *Hydrobiosis* spp., have also been found to be an important dietary item (West *et al.*, in prep.). Browsing over the substrate by banded kokopu is also commonly seen individuals kept in aquaria (pers. obs.), and these prey would no doubt be captured in much the same way as is noted with a close relative of the banded kokopu; the shortjawed kokopu *Galaxias postvectis* (Eldon, 1969; McDowall, 1996; McDowall, *et al.*, 1996). This form of prey selection would appear to be primarily by visual means as it was not an encountered component of their typical night time activity.

Shoaling in currents in streams has also been noted by Eldon (1969), and although he found that they were solitary in aquaria (possibly again because of a lack of a distinctive water current and direction) banded kokopu have been observed to shoal in the still water lacustrine environment of Lake Ototoa (South Kaipara head) (pers. obs.). This may be more due to the use of schooling as a predator avoidance behaviour in this situation, however, than for optimisation of energy and prey capture given the very different nature of feeding and prey in these lentic environs (McDowall, 1990). It is possible, given the circumstances in which banded kokopu appear to shoal together i.e.

the presence of a distinct directional current, that this behaviour may involve a streamlining function (c.f. V-formations in long-distance migrating waterfowl). Alternatively it may simply be due to maintenance of an intraspecific territorial space around each fish in the form of a hierarchy with the larger fishes dominating the optimal foremost positions, as has been noted with other salmonids (Fausch, 1984). More specific research would be required to answer this question.

The caudal fin of adult banded kokopu is typically symmetrical and quite rounded (Stokell, 1955). This morphology implies that no pitch is evolved during swimming, and that an individual's vertical position in the water column is maintained solely through the close approximation of the density of a teleost with that of the ambient water medium (Hildebrand, 1982). With an low aspect ration of only around 1.0 and a thick peduncle (Plate 0.02/.), the morphology of the caudal fin suggests better adaptation to burst swimming (Hildebrand, 1995) than the slow cruise swimming often observed being sustained in the recordings over more than 30 minutes at a time. Sustained swimming speeds of juvenile banded kokopu, however, are also equal to other comparable native fish with more forked tails such as inanga (*Galaxias maculatus*) and common smelt (*Retropinna retropinna*) (Mitchell, 1989). Moreover, their shallow body shape would classify banded kokopu as rheophilous (fast-water loving) indicating a selection for these fishes to be able to manoeuvre rapidly and easily in fast waters.

In addition to the aforementioned collapsing of the anal and dorsal fins to reduce drag, banded kokopu were also seen to utilise shelter from obstacles in the main flow water current such as large cobbles and other large substrate components (LSC). These obstacles cause a slightly lower water velocity immediately above themselves, due to an upthrust of the deeper water at this point. Use of these such instream items has also been noted with brown trout *Salmo trutta*. LSC has been found to be accountable for determining much of their position choice in water bodies (Hayes & Jowett, 1994). Additional use of backwaters, shallows and other slow water habitat adjacent to fast water has also been noted with the closely related koaro (*Galaxias brevipinnis*) (Hayes, 1996).

The fact that koura were not as common in the unnamed Coromandel stream as in Swanson Stream, may be due to both the high densities, and typically larger sizes of banded kokopu found in this former stream. Dietary analyses of banded kokopu have found that koura form a small but gravimetrically important part of the diet (Main, 1988; West, 1989; Halstead, 1994). Conversely, the aggressive behaviour seen shown by koura to banded kokopu may be interpreted as either a defensive action or as attempted piscivory, koura have been known to capture and kill banded kokopu of greater weight than themselves when in captivity (pers. obs.). Although significant numbers of large koura were found to be present in Sawmill Stream whilst spotlighting at night, no doubt the adaptation of a swimbladder and a large component of nocturnal activity helps to separate the fishes from the resident populations of actively foraging koura in most situations.

CHAPTER 3.

Nocturnal Habitat Requirements of Banded Kokopu

"At present, understanding of habitats of New Zealand's native fishes is meagre and largely qualitative…. Quantitative information is needed."

McDowall (1993)

"One of the most effective ways to learn about microhabitat utilisation by fish is to simply quietly and unobtrusively watch them."

Bovee (1986)

3.0 Introduction

Often one of the main difficulties in determining an instream environmental flow (e.g. minimum flow) requirement is a lack of quantitative habitat preference data for the species inhabiting those waterbodies (Gordon *et al.*, 1992). This paucity of data has been recognised as presenting a significant dilemma in New Zealand waters when reduced flow issues arise (Church *et al.*, 1979; McDowall, 1993). When these data are known, the IFIM (Instream Flow Incremental Methodology) process can then evaluate the effects of changes in stream flow on hydraulic conditions, water temperature, water quality, etc. in order to finally recommend a flow regime that will lend itself to the most desired stream biology (as a function of species and their expected resultant biomasses given these stream determined conditions) given further, anthropogenic, constraints. Habitat preference data for flowing waterbodies is therefore vital in order to assess flow minimum requirements for fish and other cohabiting instream species (Bovee, 1986).

Understanding the distribution and preferences of freshwater fishes allows development of models e.g. PHABSIM (Physical HABitat SIMulation system) that enable prediction of the impacts of changes in specific relevant environmental variables on habitat availability and fisheries abundances (McDowall, 1993). Habitat modelling on the basis of just a few basic habitat parameters provides freshwater fisheries managers with a further powerful tool that allows the impacts of water management strategies (e.g. stream water abstraction, and its converse stream water retention) to be predicted. Further they supply fundamental data for more complex flow assessments on the basis of habitat to provide a rational and defensible approach to the allocation of water by taking into consideration the hydraulic response of the river to changes of flow and the potential effect on biota (Ian Jowett, National Institute of Water and Atmospheric Research, pers. comm.; Gordon *et al.*, 1992).

The assumption that abundance of a given freshwater fish species at a given site is likely to reflect habitat suitability has potential to be unsound where a migratory life history is present, and therefore a significant additional influence to population size that may not be able to be modelled. However, wherever a species readily occurs it is likely to occupy the most suitable habitat available to it there. As long as numerous locations are used to derive habitat suitability data, and as long as the habitat areas measured bear a good relationship to the species' occupied niche, summation of the data from many localities is likely to reflect that species' habitat preferences and to provide general fundamental data for habitat modelling. The concept therefore underlying the "habitat preference" or "habitat suitability" function is simply that if an organism is found in a higher proportion in the conditions of a particular environment, compared to the total availability of those conditions, then it has actively selected for that set of conditions. If, however, the organism's frequency distribution is identical to the distribution of environmental conditions, then it is considered to be randomly distributed, with all measured parameter states are satisfactory for the species. The relative preference of a given life stage of a given species is therefore simply defined as the ratio between utilisation and availability (Bovee, 1986) i.e.:

$$P_r = \frac{P[E/F]}{P[E]}$$

where; $\mathbf{P}_r$ = the relative preference of a given life stage of a given species for a specified range of environment conditions.

P[E/F] = the probability of a specific habitat variable being utilised by the organism (indicated by the proportion of its presence at this variable during sampling, over the number of such samples taken).

P[E] = the probability of the given habitat variable occurring in the waterbody during the time in which the habitat usage data was sampled.

A fundamental key to successful flow recommendations is therefore to provide sufficient suitable habitat or the maintenance of all life stages of the target species (Jowett, 1996a). As a consequence of this information's significance, habitat preference curves have already been derived for many of native freshwater invertebrates (Jowett & Richardson, 1990) and fish species (Davis *et al*., 1983; Main, 1988; Taylor, 1988; Hayes & Jowett, 1994; Jowett & Richardson, 1995; Hicks & Barrier, 1996). These data, in addition to a valuable contribution to the understanding of the pure ecology of the species, are also clearly of further great value in the more general management of the described species. However, none of these studies have examined what (if any) differences there may be present between the habitat preferences any of the major life stages (e.g. juveniles and adults), as has been investigated and analysed with some overseas analyses (e.g. Jowett, 1996c; Hicks & Hall, in prep.).

Although habitat preference curves had been previously developed for banded kokopu in the South Island (Main, 1988), little work had been performed with potentially differing North Island populations of this species, and there were also several behavioural and ecological aspects which it was felt necessitated further investigation. As previously noted, no discrimination had previously been made among fish ages, with all age classes being grouped together during field data collection. Preference curves for velocity from these data revealed an optimal point at low water velocities, but what also appeared to be a paradoxical *increasing* preference at higher water velocities (Fig. 3.01/.).

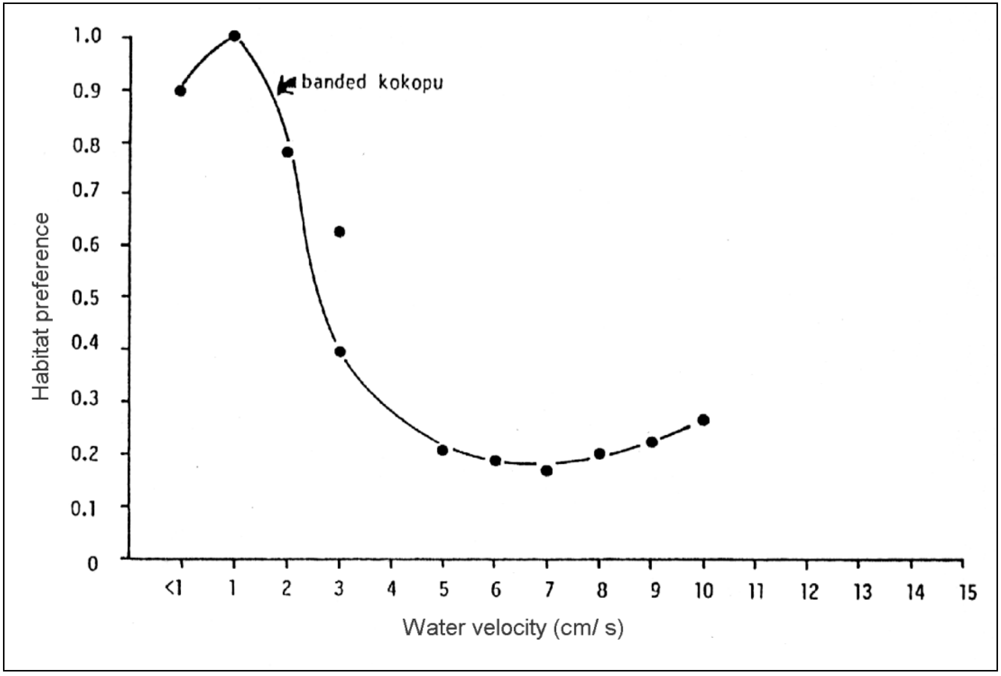

Figure 3.01/. Main's (1988) water velocity nocturnal preference curves for banded kokopu. Note the apparent growing preference at higher velocities from a lowest preference at low velocities.

It was thought that this apparent bimodal nature of this depth preference curve may be due to more than one age class (and consequent potential ontogenetic behavioural type) being considered together. Halstead (1994) also proposed that, although there did not appear to be any clear relationship between fish greater than 120 mm total length and water velocity, bigger fish appeared to predominate in the slower water (up to 0.06 m/s) with smaller fish being found in much faster flows (up to 0.23 m/s). This hypothesis was further implied by previous personal observations during the spotlighting calibration fieldwork (Chapter 1) where it appeared that smaller fish (0+) frequently occupied a different habitat type to larger fish (1+) (*Ibid*). These smaller fishes typically occurred in the faster, and often shallower waters within the same given streams. Indeed it is not novel for different ages of a given fish species to have different habitat preferences e.g. Hayes & Baird (1994).

Furthermore, data for substrate preferences was also not collected in this study; a variable which, whilst not as significant for IFIM usage as depth and velocity, is still very much desirable for assisting in the management of a species' fishery and for stream rehabilitation. Sampling of availability data was also thought to have been possibly biased by the non-random systematic methods used, with sampling having occurred at consistent distances along the stream bed both longitudinally and transversely. A more purely random methodology for determining available habitat was thought to be an improvement on this previous methodology, being less biased in nature. In addition to this, data collected for cover preferences did not include how often fish utilised habitat with no cover, with a consequence that no judgement could therewith be made on the importance of this habitat variable in the light of this lack of quantitative information.

Electric fishing was considered to be both less efficient (see Chapter 1) and less accurate than spotlighting at night, as banded kokopu were often seen to flee ahead of the operator during their attempted capture in electric fishing, indicating that they were not being caught in their pre-disturbance utilised habitat (pers. obs.). Following on from McDowall's (1993) suggestion that seasonal timing of the study of habitats is very important to ensure that a species' habitat preferences are measured at a time when the species occupies the relevant study habitat, it was also thought reasonable to also consider the time of the day when the species is most active in its habitat (Orth, 1987). Observations of the closely related koaro (*Galaxias brevipinnis*) outside of the typical daylight electric fishing times have already shown this species to utilise habitats previously found to be unassociated with its diurnal use (Hayes, 1996). Thus, as had

Main (1988) also chosen, spotlighting was therefore selected as the method with which to determine the positions which banded kokopu occupied (habitat “used”) at night. Additionally, habitat use surveying took place at times in the evening where there was expected to be a peak of banded kokopu activity (Chapter 2, Fig. 2.02/.), yet with a sufficient paucity of sunlight such as to enable the fish to be both readily approached and observed.

3.1 Methods

Data collection

Over a series of nights in the months of October and November 1997, the nocturnal habitat use by banded kokopu and their available habitat was sampled late in the evening in streams in the Coromandel Ranges (Sawmill Stream and Taumatawahine Stream), West Auckland (Swanson Stream), and the Hakarimata Ranges (Waipuna Stream). These streams were chosen firstly on the basis that it was known that they had high fish densities, such that preferred microhabitats should be occupied, and secondly that with banded kokopu known to be diadromous that there were no known migratory obstacles downstream (McDowall, 1993).

Working in an upstream direction along the stream banks, the positions of undisturbed individuals of banded kokopu were identified and a marker (Plate 3.01/.) placed directly underneath this position on the substrate underneath. The positions of disturbed fishes were not marked but instead ignored, as their position may have been due to an escape response rather than a preference to the characteristics of their location in the stream.

Plate 3.01/. The Spanish habitat use marker model (right) which the New Zealand version (left) was based on.

The great majority of fish did not appear to be disturbed when approached; and although there was a difficulty at times accurately identifying the smaller banded kokopu 0+ fishes from the inanga juveniles also present in Lawlor Stream. This was thought to be of little consequence as more than half of a 40 fish sample of these 0+ fishes were found to be banded kokopu through removal to aquaria for later development (Paul Woodard, Native Freshwater Fish Society (NZ), pers. comm.). Markers could be readily placed directly at the focal point of the still stationary banded kokopu, allowing accurate identification of the physical parameters at this point to be noted in what is typically a highly heterogeneous habitat (Plate 3.02/.).

Plate 3.02/. A habitat usage marker placement placed at the focal point kokopu of an adult banded in Taumatawahine Stream, Coromandel.

Individuals were classified into two main groupings; 0+ (total lengths exclusive to 80 mm), and 1+ (fishes above this size) as with Chapter 1. This information was noted next to the number of the tag on all which was recorded on pre-printed habitat use datasheets (Appendix (D), Tables 5.04/., 5.05/.).

The following day, given that no rain or other hydrologically significant events had occurred, the depth, substrate size (Table 3.01/.), and both surface (only for 3 of the streams surveyed) and average (0.6 of the total column depth from the surface) column velocities were noted for each of the kokopu positions found during the previous night (these previous positions as indicated by the markers present). Surface velocities were noted due to the relevance of this variable given that this species is though to feed primarily at the water's surface on terrestrial allochthonous material. Focal-point (snout) velocities were not noted, as in addition to the difficulty in estimating fish depth in the water column by spotlighting, focal-point velocities are not suited to many existing hydraulic models (Bovee 1986). Cover within 0.6 m of the marker's position was noted as being "used" habitat, as was also significant cover afforded by terrestrial vegetation overhanging the surface of the water from the bank to within 0.6 m. All velocities and gaugings were measured with a Marsh & McBirney model 201D portable water current meter.

Plate 3.03/. Francisco ("Paco") Martínez Capel taking a measurement of used habitat with a water velocity meter. Taumatawahine Stream, Coromandel.

Table 3.01/. Substrate size classes used and their descriptions.

Description	Length of intermediate axis (mm)
Silt/ mud	< 0.062
Sand	0.062 - 2
Small gravel	2 - 8
Large gravel	8 - 64
Cobble	64 - 264
Boulder	> 264
Bedrock	Continuous rock

Random points were determined by dividing the largely even width reach by the number of usage samples, and then sampling once along a random cross-section at each of these systematic points (Rinne, 1985). Therefore, for a 100 m reach with the positions of 67 kokopu the previous night, a single sample would be taken at a cross-section at each sequential 1.5 m (100 m/ 67). Upon reaching this cross-section, the actual random habitat availability point position was determined via the proportion of a 2-digit random number generation from the true right bank. For example for a 1.6 m wide cross-section the randomly generated number 85 would indicate the point to be sampled was 85%, or 1.36 m, from the true-right bank. The same variables as had been previously noted for the nocturnal kokopu positions were again measured for the random samples of habitat availability i.e. depth, substrate, cover, and surface and average column velocities from these points. Finally a gauging was made of the stream at the downstream point where nocturnal usage sampling had begun.

Lengths of the 3 main habitat types; pool, run, and riffle had been noted in the 20 m stream reaches electric fished during the calibration of the spotlighting method. Average reach widths had also been noted from at least 5 measurements as were the numbers and lengths of banded kokopu captured. Streams included in this habitat mapping were both the Swanson streams, Sawmill Stream, and the Waipuna Stream (a total of 16 data points).

Data analysis

Available and used data for velocities, depth and substrate were analysed on a PC using the computer program “Kernel” (Aptech Systems Inc. 1992). The nominal classes of cover were analysed using simple normalised utilisation/ availability ratios (Bovee 1986) via a spreadsheet after Hicks & Barrier (1996). Data for 1+ and 0+ fishes were analysed separately and then together to produce single variable curves for all fish. Although previous works have, e.g. Main (1988), cover preference data was not fitted to curves, due to the nominal nature of this variable, but was instead displayed as a histogram.

Although minimal smoothing of habitat use and availability curves was used to retain as purest original information as possible, greater smoothing was required for habitat use curves than for habitat available curves. These latter curves were typically more even in nature such that smoothing was therefore first attempted with used habitat curves. Whilst a maximum smoothing factor of 2 was used for most data, depth curves were particularly erratic and required a smoothing value of 6 for their usage curves. Suitability functions were then fitted to these smoothed curves and these data finally entered into the relevant "RHYHABSIM" IFIM modelling program's directory (Natives.prf).

3.2 Results

1+ fish were more abundant in the streams than 0+ fish, with only 87 data points being realised at the end of the study for 0+ fishes compared to 204 for 1+ fishes (Table 3.02/.). Altogether a total of 291 used habitat and 285 available habitat data points were accumulated from five separate streams in 3 different regional locations (West Auckland, Central Waikato, and the Coromandel Ranges). Stream morphology ranged from the cobbly, steep, and turbulent stream of the Taumatawahine Stream; to the narrow and deep sandstone-cut Swanson Stream with its lack of coarse substrate and modified riparian; and the hard greywacke, pristine forested Waipuna Stream (Plate 0.03/.).

Table 3.02/. Sources and sample numbers of data used to determine nocturnal banded kokopu habitat preference curves.

Stream name	Date (night/ day)	Number of used points		Number of available points	Gauging (m^3/ s)
		0+ fish	1+ fish		
Lawlor Stream	25 & 26/10/97	39	8	52	0.0089
Swanson stream	26 & 27/10/97	14	31	50	0.0025
Taumatawahine Stream	3 & 4/11/97	14	55	70	0.0057
Sawmill Stream	4 & 5/11/97	2	76	74	0.0028
Waipuna Stream	10 & 11/12/97	18	34	39	n/d
Total		**87**	**204**	**285**	
Total all ages		**291**		**285**	

n/d = no data available for this stream.

To determine whether significant correlations had occurred with the banded kokopu preference data variables (clearly water velocity and depth are frequently correlated) (Jowett & Richardson, 1994) a Pearson's product-moment correlation analysis of the combinations of variables sampled for was executed. Habitat variable combinations (mean column velocity/ column depth; substrate size/ mean column velocity; substrate size/ column depth; surface velocity/ mean column velocity) revealed no statistically significant correlations (Table 3.03/.). Available mean column velocities and surface velocities, however, were found to be highly significantly, positively correlated with each other in an almost direct manner.

Table 3.03/. Significance of correlations between habitat variables measured; *N*=284 (**N*=185).

Variable correlation	R_s, R^2	P (two-tailed)
Mean column velocity/ column depth	0.298, 0.089	<< 0.001
Substrate size/ mean column velocity	0.199, 0.039	<0.002
Substrate size/ column depth	0.179, 0.032	< 0.005
Surface velocity/ mean column velocity*	0.949, 0.901	<< 0.001

Habitat preferences for all fish

Preferred mean velocities for all fish sizes (Fig. 3.02/.) were within a very narrow range, with an optimal velocity of only 0.01 m/s. The preference range included slow negative (back current) velocities in addition to slow positive velocities. The preferred surface velocity range was similar (-0.05 to 0.05 m/s) with an optimal velocity also of 0.01 m/s (Fig. 3.03/.). As with mean column velocities, most water velocities available were in a slower range (-0.10 to 0.30 m/s). Preference for depth increased with increasing water depth to a depth of around 0.67 m, and then began to quickly decrease with further increases in water depth; depths below 0.40 m were not very suitable (Fig. 3.04/.). Most available habitat was shallow (less than 0.20 m deep). There was very little distinct preference shown for substrate size (Fig. 3.05/.), although the optimal substrate type was found to be silt and mud. Generally decreasing preference was shown for increasing substrate size classes, although bare bedrock was shown to elicit an increased preference over boulders. Density of available habitat was relatively normally distributed among the size classes (note: not a linear scale) with a predominance of fine and coarse gravels (2-64 mm). Preferred cover (Fig. 3.06/.) was from the roots of riparian trees and undercut banks, although tree ferns (primarily *Cyathea* spp.) also appeared to be important items for cover. Cobbles and other substrate, overhanging vegetation, and instream debris also displayed high preferences; although a notable preference was also shown for no cover being present (preferred value of 0.74). There did not appear to be a large amount of cover available in the streams surveyed, with a density of 0.65 for no cover being present.

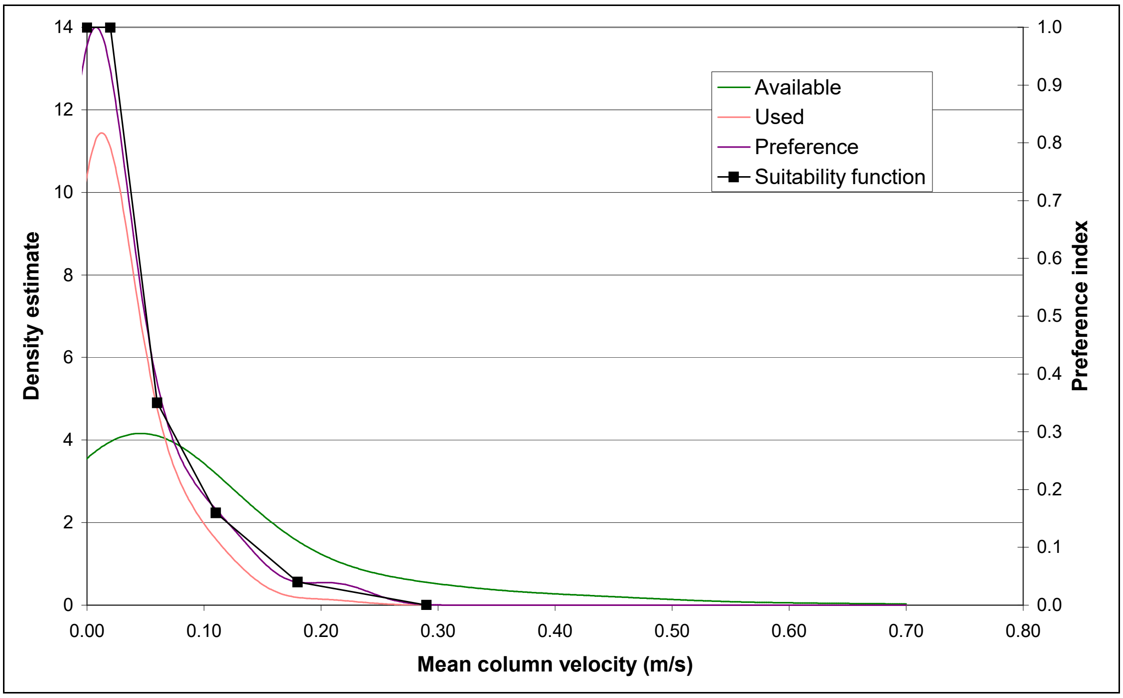

Figure 3.02/. Density estimates of used and available habitat for all sizes of banded kokopu for mean column water velocity. The derived preference curve and suitability function are plotted on the secondary x-axis.

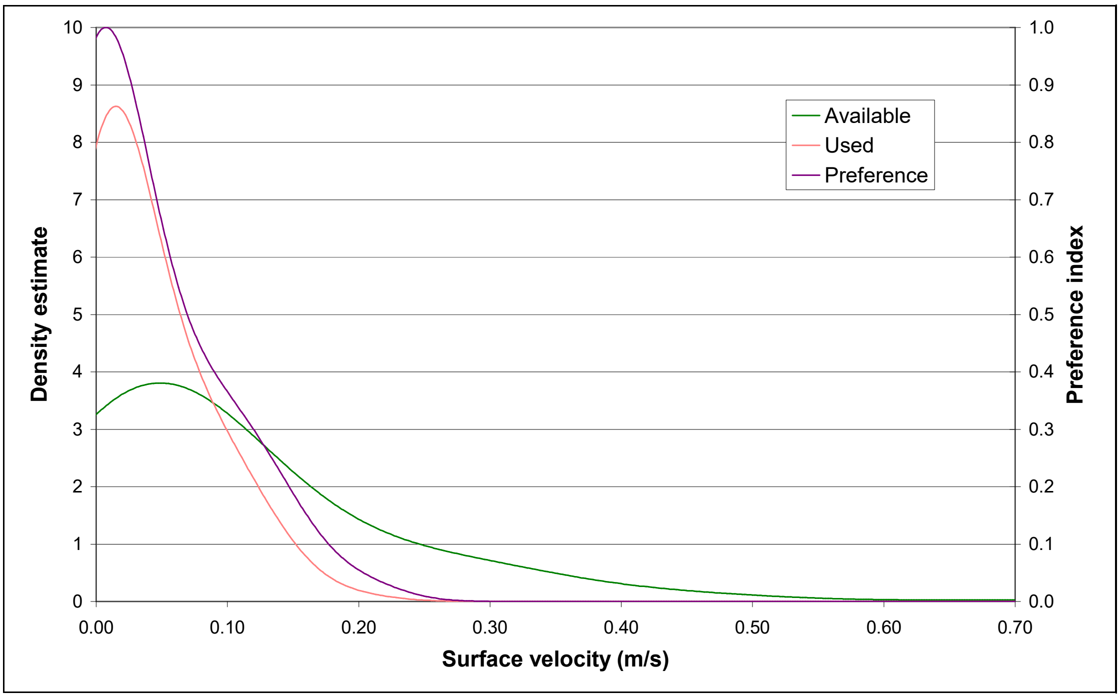

Figure 3.03/. Density estimates of used and available habitat for all sizes of banded kokopu for surface water velocity. The derived preference curve is plotted on the secondary x-axis.

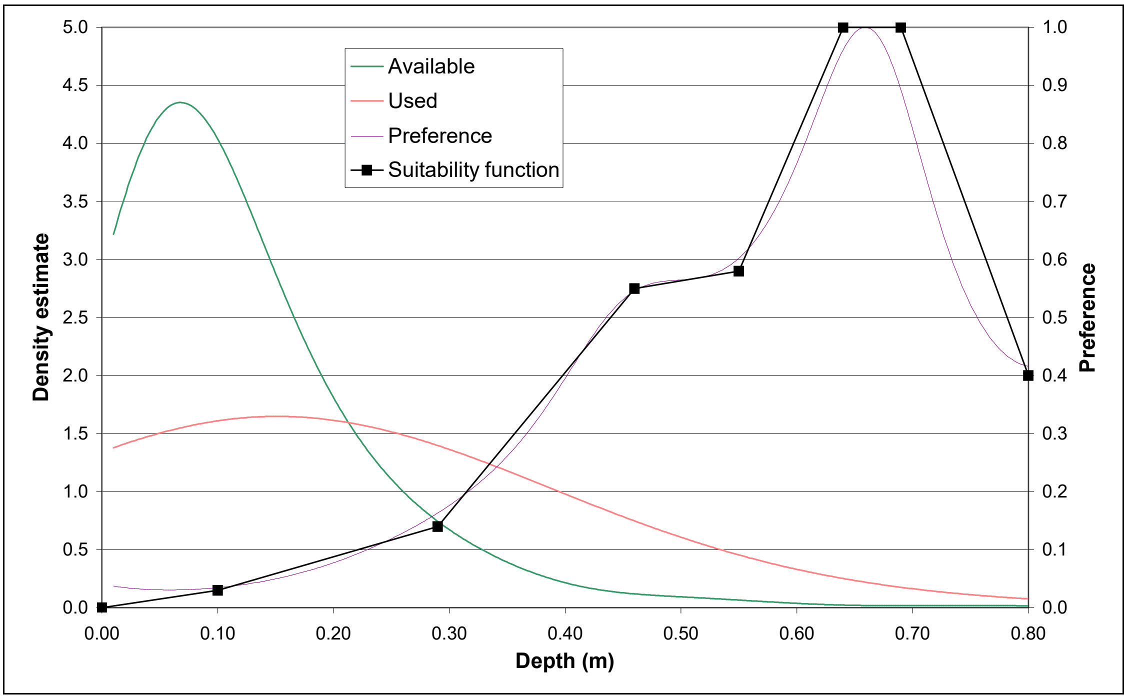

Figure 3.04/. Density estimates of used and available habitat for all sizes of banded kokopu for water depth. The derived preference curve and suitability function are plotted on the secondary x-axis.

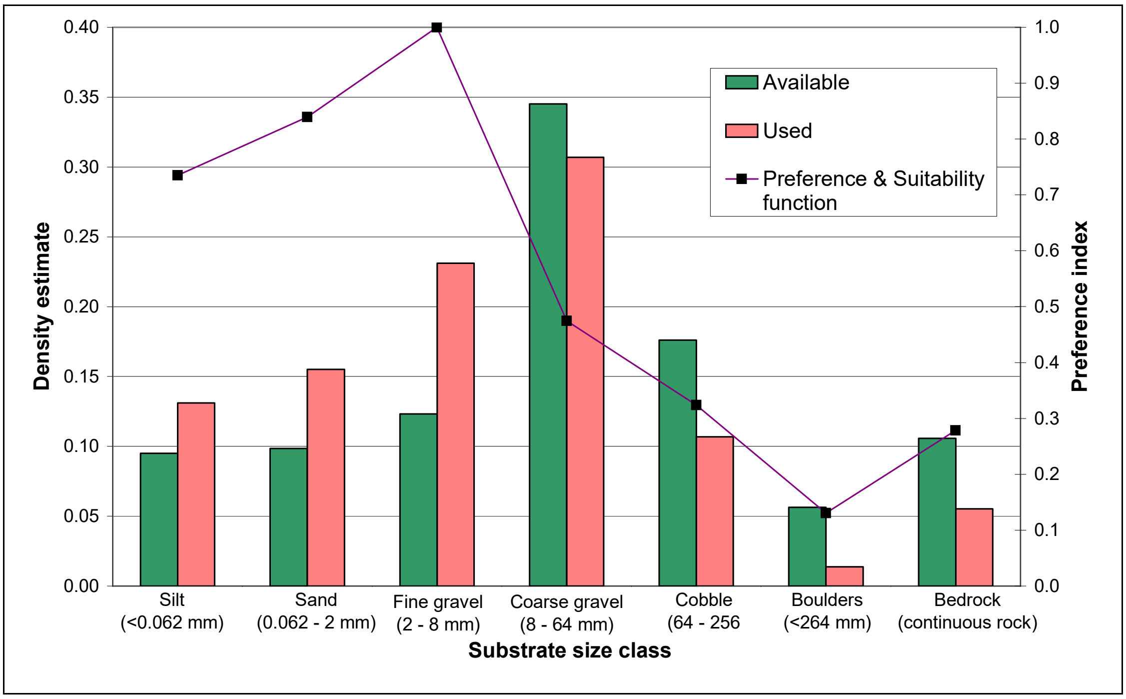

Figure 3.05/. Density estimates of used and available habitat for all sizes of banded kokopu for substrate size. The derived preference curve and suitability function (same line) are plotted on the secondary x-axis.

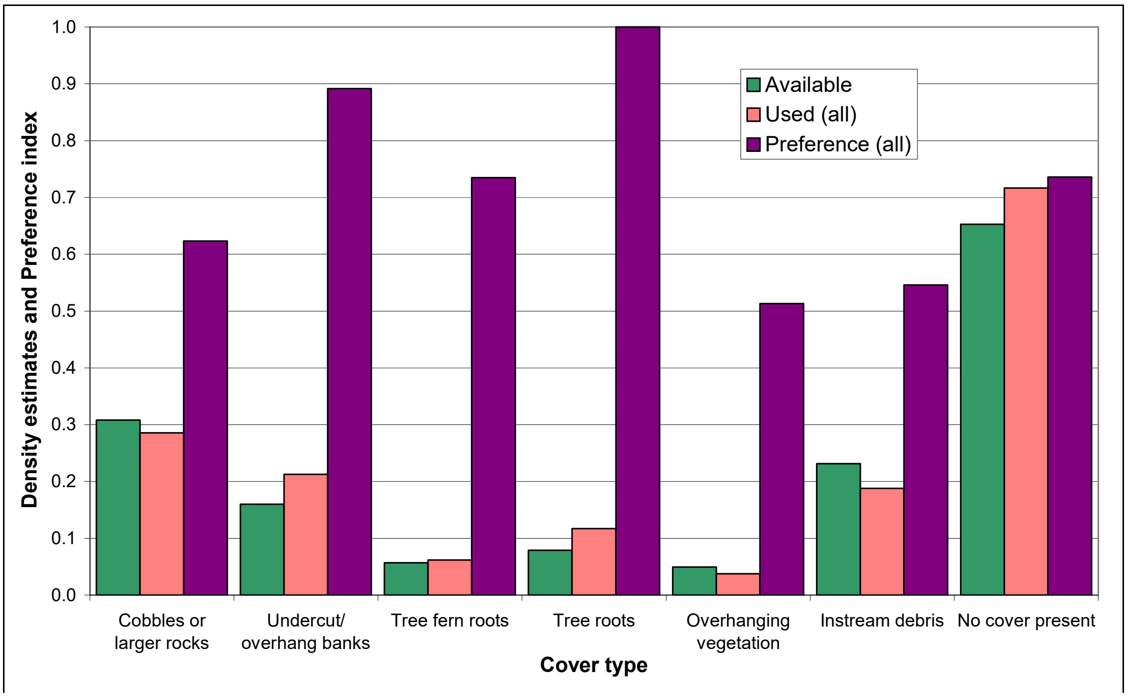

Figure 3.06/. Density estimates of used and available habitat for all sizes of banded kokopu for cover type. The derived preference curve and suitability function (same line) are plotted on the secondary x-axis.

Habitat preferences for 1+ and 0+ fish

Preferred mean column water velocities were slightly higher for 0+ fish than for 1+ fish (0.01 and 0.05 m/s respectively) (Fig. 3.07/.) and both occupied narrow (albeit marginally overlapping) preference ranges. Areas with slightly negative water velocities (to – 0.03 m/s i.e. slow backwaters) appeared to be also shown a strong preference by 1+ fish, although were not used by 0+ fish who inhabited water velocities only as low as 0 m/s. 1+ fishes did not appear to utilise waters with velocities over 0.05 m/s; although 0+ fishes had a broader preference range up to around 0.10 m/s. Habitat preferences for surface water velocities were almost the identical to those for the mean column water velocities, with 0+ fishes again preferring slightly faster surface waters than the 1+ fishes with their slightly reduced velocity preferences (only 0.08 m/s). 1+ fishes preferred velocities were the same for the surface as they were for the column mean at 0.01 m/s. These preferred velocities for separate 1+ fishes and 0+ fishes were not noticeably different from the aforementioned preferences for all fish ages. As with mean column velocities, most water velocities available to the fishes were in a slower range (-0.10 to 0.30 m/s).

Preferred water depths for 1+ fishes closely mirrored those of the 0+ fishes, with optimums at around 0.5 m deep (Fig. 3.08/.). 0+ fishes may have begun to show a dislike for water depths greater than 0.7 m although there was little availability data to demonstrate this trend further as most available habitat was shallow (the majority of the streams were less than 0.30 m deep). Preferred substrate sizes for 1+ fishes again closely mirrored those for the 0+ fishes, with optimums in the smaller size classes (silt and mud) (Fig. 3.09/.). With very broad preferences shown, optimal size classes were very indistinct. These preferences were therefore clearly very similar to the preferences also shown by fish grouped by all ages. Preferred cover of 0+ fishes was approximately similar to that of 1+ fishes, although tree fern roots appeared to be of much greater importance to the latter (Fig. 3.10/.). Cobbles and other substrate types were also used much more often by 0+ fishes than by 1+ fishes. As with the all sizes, notable preference was also shown for no cover being present (preferred value of 0.73 - 0.74) and there was not a large amount of cover available in the streams surveyed, (a density of 0.65 for no cover present).

It is clear that the preference curves for all fish closely resembles those of the 1+ fishes only curves. This is not surprising given that the all fish preference data was heavily weighted toward 1+ fishes' preferences due to the large proportional component of 1+ fishes' habitat usage data (more than 70%) (Table 3.02/.).

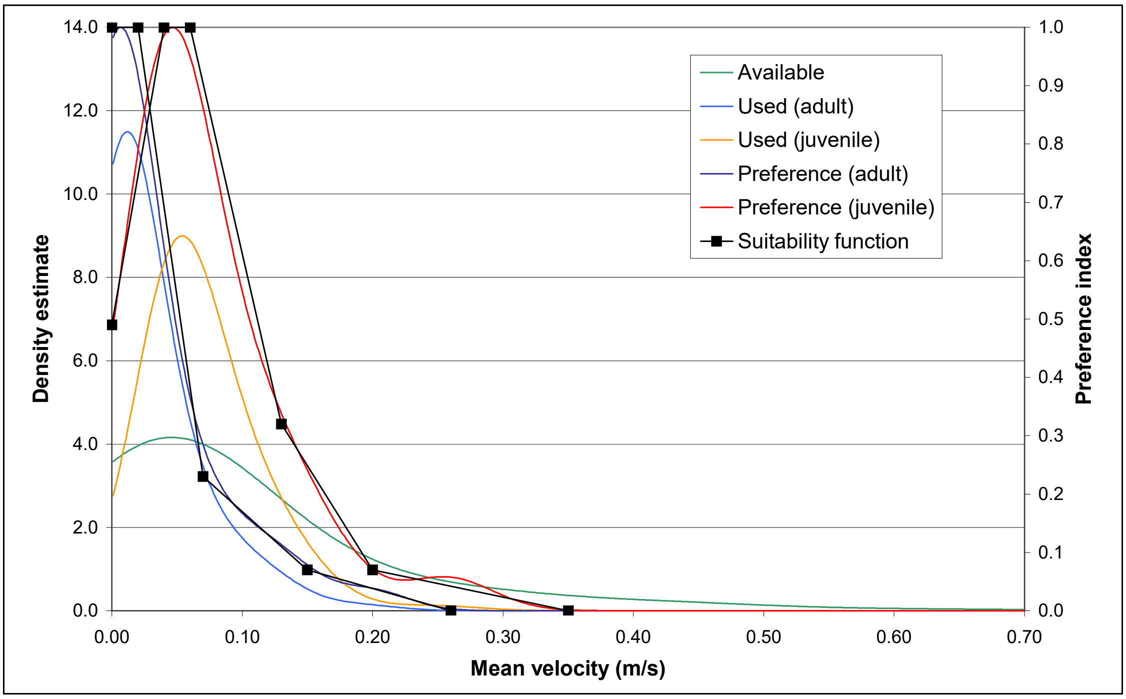

Figure 3.07/. Density estimates of used and available habitat for 0+ and 1+ sizes of banded kokopu for mean column water velocity. The derived preference curves and suitability function are plotted on the secondary x-axis.

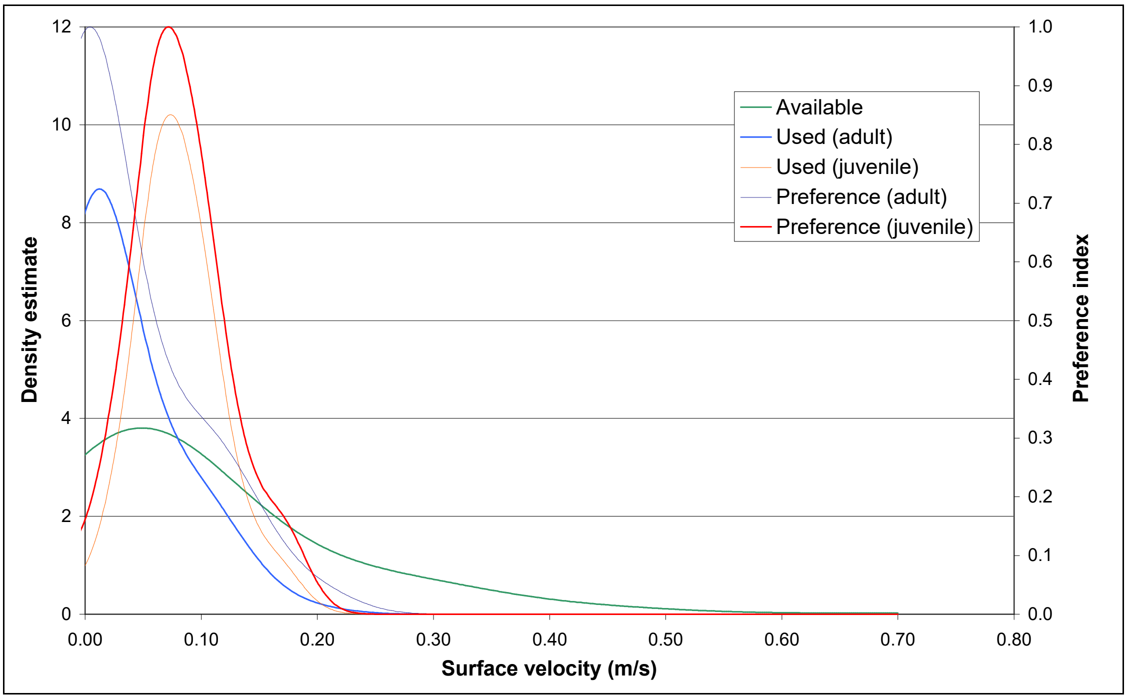

Figure 3.08/. Density estimates of used and available habitat for 0+ and 1+ of banded kokopu for surface water velocity. The derived preference curves is plotted on the secondary x-axis.

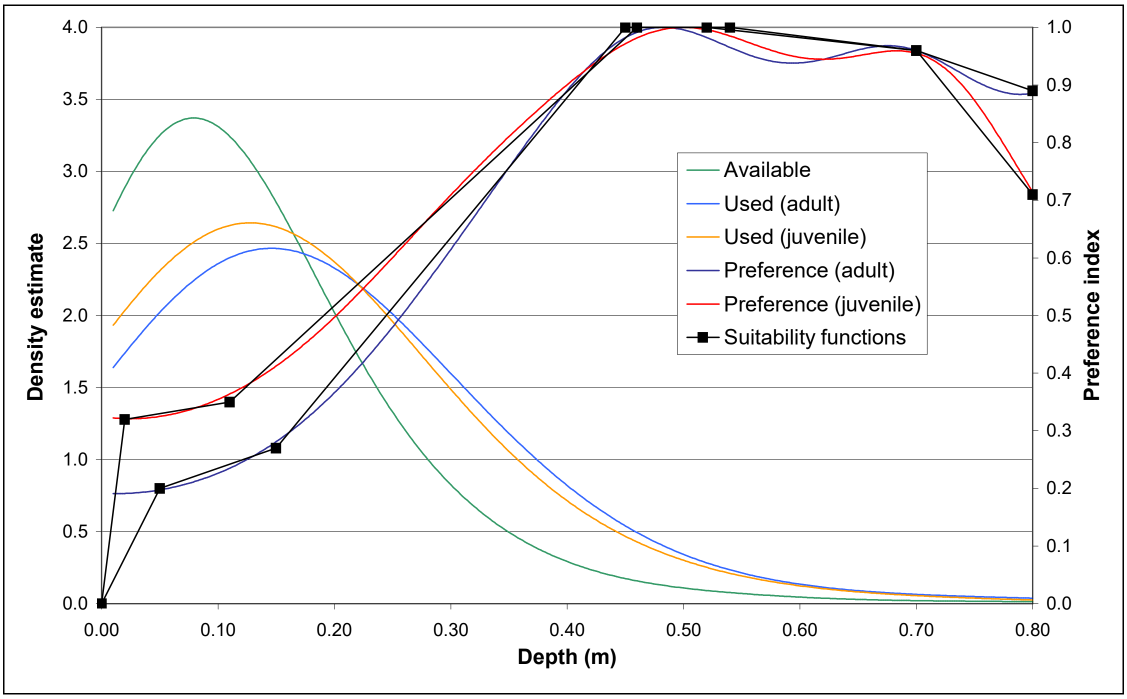

Figure 3.09/. Density estimates of used and available habitat for 0+ and 1+ sizes of banded kokopu for water depth. The derived preference curve and suitability functions are plotted on the secondary x-axis.

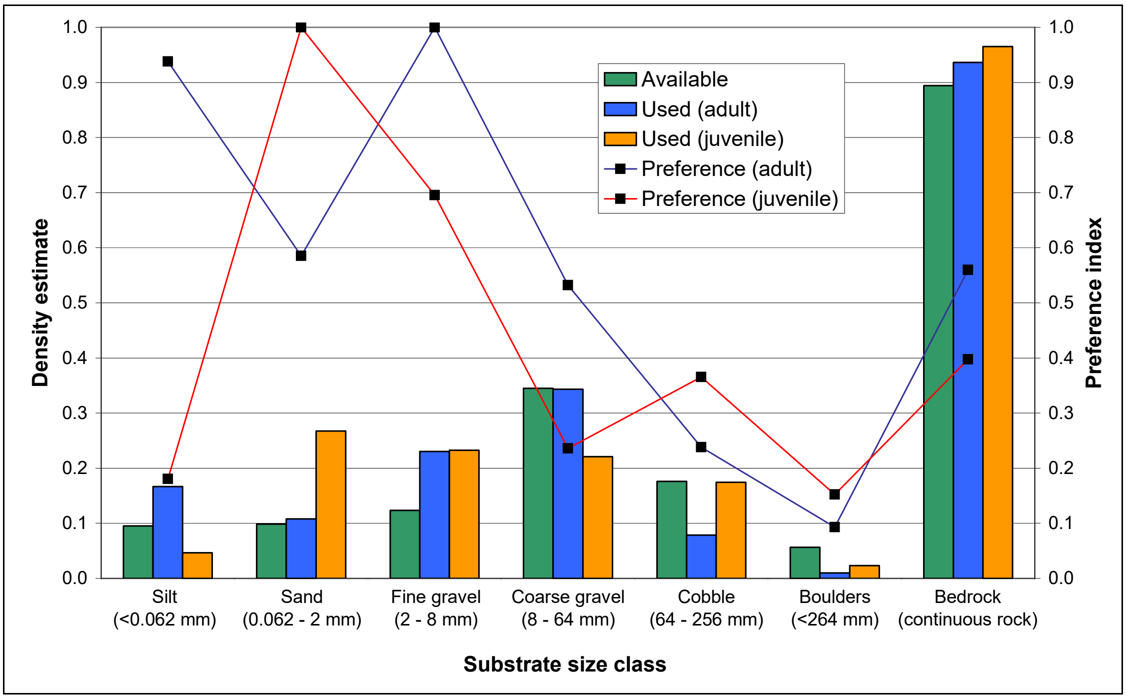

Figure 3.10/. Density estimates of used and available habitat for 0+ and 1+ sizes of banded kokopu for substrate size. The derived preference values and suitability function are plotted on the same lines on the secondary x-axis.

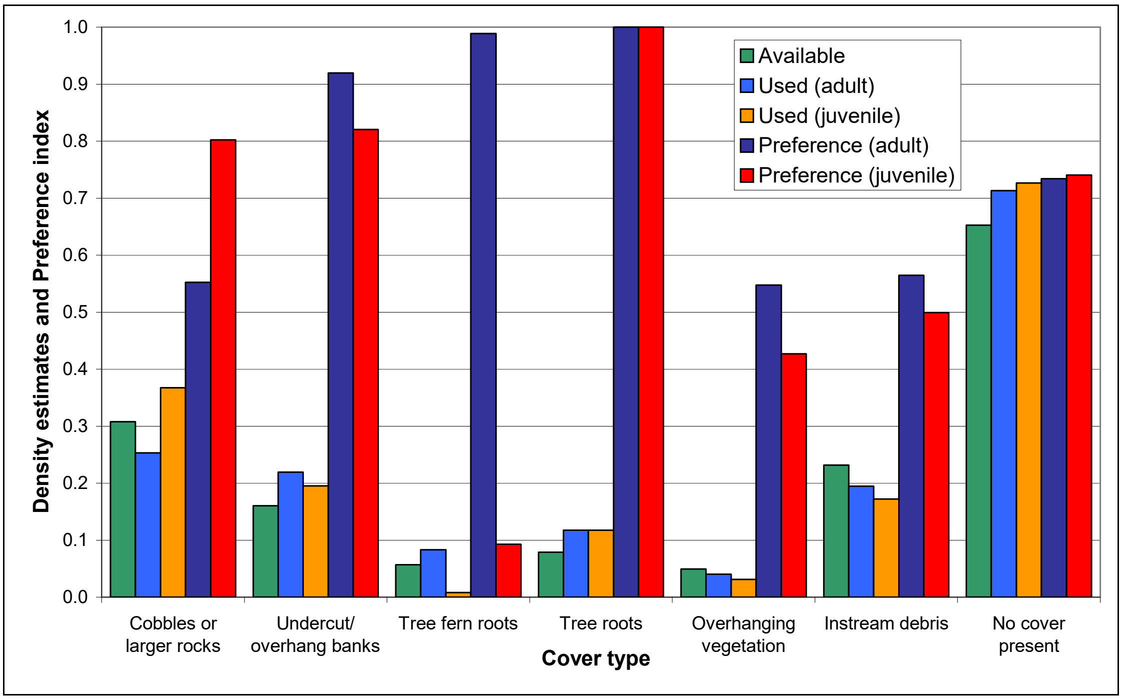

Figure 3.11/. Density estimates of used and available habitat for 0+ and 1+ sizes of banded kokopu for cover. The derived preference values are plotted on the secondary x-axis.

No significant correlations were noted between the surface area of pools, surface area of runs, and the surface area of riffles present in the 16 20 m reaches measured. No significant correlations were also noted between the surface area of these pools and riffles and banded kokopu abundances for either 0+ fishes, 1+ fishes, or all sizes together (Table 3.03/.). A single highly significant correlation was, however, noted between abundances of banded kokopu of all ages and the surface area of pools. Neither 0+ fishes or 1+ fishes alone showed significant correlations for abundance with this habitat type.

Table 3.04/. Relationship of 20 reach surface area (m^2) of main habitats types to banded kokopu abundances. n/s = not a significant correlation at the $\alpha = 0.05$ level or less. *N* = 16.

Habitat	**All sizes**		**0+ fishes**		**1+ fishes**	
	r^2	**P (2-tailed)**	**r^2**	**P (2-tailed)**	**r^2**	**P (2-tailed)**
Pool	0.014	n/s	0.100	n/s	0.055	n/s
Run	0.031	n/s	0.010	n/s	0.014	n/s
Riffle	0.386	< 0.01	0.039	n/s	0.137	n/s

Whilst sampling for habitat use in the Waipuna Stream in December, very small banded kokopu 0+ fishes were observed under spotlight, with total lengths as low as 35 mm. Given that the inland distance of the Waipuna Stream population from the sea is around 100 km, it seems likely that this population is lacustrine (probably completing their life-cycle in Lake Waahi only *c.*14 km downstream) as has been previously proposed but not conclusively proven by Gumbley (1993).

3.3 Discussion and Conclusions

As has previously been noted when using spotlighting to estimate the abundance of banded kokopu in streams, the ability to detect banded kokopu by spotlighting at night is characteristically influenced in a negative manner by the presence of a broken water surface, and also by depth (*ibid.*). Whilst depth is unlikely to be a significant source of error in the preference curve data selection due to the typically shallow nature of the streams sampled (the majority less than 0.30 m deep) (Plate 0.03/., Fig.'s 3.04/., 3.09/.)

broken water is yet a very real source of bias toward less observations in fast broken water i.e. riffles.

Substrate, depth, and velocity are often inter-related under natural conditions and therefore patterns observed may not imply a particular preference, but instead may merely reflect the association of in-stream hydraulic factors (Jowett & Richardson, 1990). For example, from his work in the Hakarimata Ranges, Hanchet (1990) suggested that streams in forested land generally had much greater instream cover from woody debris, a lower substrate coarseness, and also lower water temperatures than those below forest. This presents a very real problem with habitat preference analysis, such that it may not be possible to determine which of these variables are correlated and which are not; all possibly being merely correlated with a key variable (Rowe, 1991). Due to the lack of significant correlations found between the variables depth, mean column velocity, and substrate size, this issue would not appear to be a problem in this study. Surface velocities, however, would largely appear to be a reflection of their mean column velocities.

At least for one other species (brown trout, *Salmo trutta*) it has been suggested that despite the many practical advantages of using mean-velocities in habitat modelling, it is still recommended that focal-point velocities be used preferentially in habitat management applications as these fish were closely oriented to the benthos (DeGraff & Bain, 1986). Mean column velocities may also be more appropriate as they include a measure of the faster water above the fish (Hayes & Jowett, 1994) in which they also feed (*ibid.*, Chapter 2). In this situation with the surface oriented banded kokopu, mean column velocity was very strongly correlated with the surface velocity ($R^2 = 0.95$) such that mean column velocity was therefore a good estimate of useable habitat.

Although banded kokopu have been found to be able to readily detect and capture surface prey items at drift speeds of over 0.25 m/s, its typical habit of slow running waters (Main, 1988; West, 1989; McDowall, 1990; *ibid.* this Chapter) may instead be related to energy expenditure during sustained swimming in the flow. There would also be an increased ability to acquire prey in slower water given the longer time it would take for the prey item to pass (Halstead, 1994). Other species e.g. rainbow trout (*Oncorhynchus mykiss*) are also often found in energetically profitable resting positions of relatively low velocities, from which they then make short feeding forays into adjacent faster waters, with the consequent suggestion that maybe a single measure of velocity may be insufficient to describe a surface drift-feeding habitat (Hayes, 1994). This is also seen with kokopu utilising instream flow obstacles and also often being found near the edges of pools; possibly because of their close proximity to undercut banks and other cover found here, but also probably because of the greater proportion of backwaters and current obstructing debris more commonly present along the stream edges. To encompass this form of habitat selection, a research approach involving sampling of available habitat from a predetermined distance around the fish may be more appropriate than the systematic random methodology used in this study.

Banded kokopu have frequently been noted as having a preference for pools (Woods, 1963a; Graham, 1974; Halstead, 1994), with Hanchet (1988) even finding that pool volume was a good indicator of banded kokopu abundance, with small pools being preferred. Both 0+, 1+, and all sizes of banded kokopu can be seen to have preferences for depths and velocities as within the ranges defined by the objective criteria of Jowett (1993) as being typical of pools.

This study has also shown banded kokopu to have a very narrow and strong preference for slow waters; with 1+ fishes being absent from waters faster than around 0.10 m/s whilst 0+ fishes were found in velocities to 0.20 m/s. This phenomenon has also been noted in a previous study encompassing nocturnal behaviour where, although with a small sample size (N = 15), and not being statistically tested for, an apparent positive correlation was observed between fish size and water velocity (Halstead, 1994). In densely populated streams therefore, it may be expected to find the larger fish in the slower (and probably more optimal) microhabitats of the stream. This may cause displacement of the smaller, younger individuals by the apparently commonplace intraspecific aggression that has already been noted by both this study (Chapter 2) and another (Halstead, 1994). Furthermore to this aggression being noted, cannibalism of 0+ fishes by 1+ fishes has also been recorded during dietary sampling on more than one occasion (West *et al.*, in prep.). Preference of slightly higher water velocities in the densely populated streams which were surveyed may therefore be a reflection of a further spatial segregation in addition to the aforementioned apparent temporal segregation behaviour (Chapter 2) being demonstrated. Existence of this strategy might be detected by first measuring (or using those described here) habitat preferences for 0+ banded kokopu in streams with dense 1+ fish populations, and then reassessing these preferences following removal of 1+ fishes and the possible establishment of habitat types previously unavailable.

Given the aforementioned intraspecific aggression (Chapter 2) and further noted cannibalism (West *et al.*, in prep.) instream position choice may therefore be further constrained by the possible formation of intraspecific hierarchies in which the larger dominant fish hold optimal positions characterised by a combination of the physical variables studied i.e. depth, velocity, etc. (Fausch, 1984). This possibility therefore directly violates an assumption of habitat availability modelling; namely that habitat availability data points must be available to the fishes, clearly which they are not if this (possibly preferred position) is currently inhabited by a cannibalistic larger member of their species. Further bias may also arise from 0+ fish avoiding habitats with high density communities of 1+ fish through detection of pheromones from these larger fishes (Cindy Baker, University of Auckland, pers. comm.), which will consequently further result in low recruitment to these areas. An extension of this concept has further implications potentially to the majority of previous habitat preference studies as, even although evidence for spatial intraspecific segregation may not be perceived, *interspecific* interactions between conspecifics and *other* species present in the survey habitat such as predators e.g. eels (*Anguilla* spp.) are also likely to confound the resulting data. These interactions should therefore be a consideration in study design and analysis where possible (Orth, 1987). Although these extraneous variables are admittedly very difficult to control for in the field, there acknowledgment as a significant and confounding factor was absent from many of the habitat preference studies published in the literature. Many of these studies have been derived from water bodies with notably diverse fishery communities e.g. Davis *et al.* (1983), Jowett & Richardson (1995), etc. and this is probably simply a consequence of greater surveying efficiency, given that more species are sampled for and thus less sites are required. This used-habitat bias may also occur with only a single species present but with intraspecific predation (cannibalism) (*ibid.*) being likely (Main 1988).

Juveniles have also often noted as often being more abundant in higher-order streams (Malcolm Main, Canterbury Regional Council, pers. comm.; pers. obs.) such as 2^{nd} or 3^{rd} order streams, where higher water velocities and depths are typically found due to larger flows. Furthermore, they are often absent from many lower stream reaches after the whitebait season is finished (McCullough, 1997; pers. obs.). The especially streamlined narrow profile of the 0+ fishes compared to the 1+ fishes (Chapter 2), in addition to increasing the surface area: body volume ratio for climbing damp surfaces, would also be better adapted to the faster waters in which these species have been found to prefer. 0+ fish also have higher zero maintenance speeds than their larger conspecifics (Main, 1988).

0+ fishes may therefore be not remaining in the immediate localities in which they have been noted, but instead merely migrating through to headwaters further upstream. Use of 0+ fish preference curves may, consequently, potentially result in a sub-optimal habitat being modelled as a consequence of the preference curve methodology finding these fish in habitats which were likely not to be their most preferred given the presence of large 1+ banded kokopu. If this appears to be the case, then clearly consideration must be given to the modelling of either just 1+ fishes or all fish only, as the presence of these older fish is likely to be continued. This persistence is through both a loss of the migration instincts and a simple inability to migrate given their lower surface area/ volume relationships, and also to the high degree of pool fidelity that is demonstrated (West *et al.*, in press).

Although many authors have described banded kokopu being most commonly found in streams with a rocky bed (e.g. McDowall, 1975; 1990; Parkinson & Cox, 1990; Anon, 1996b) this study's data would intimate the contrary, with finer substrate types being preferred (Fig.s 3.05/., 3.10/.). This previous assumption is possibly due to either subjective treatment of the preferences with the latter two authors writing from South Island (especially Westland) observations where the available habitat is possibly dominated by these coarser substrate types. Substrate availability was especially normally distributed, indicating a wide range of substrate types had been encountered and had contributed to the following analysis (Fig.s 3.05/., 3.10/.). Smooth distributions for availability data were common and implied that bias had not been shown in data collection given the even distributions encountered when plotted.

Although a possible positive correlation was also noted by Halstead (1994) between water depth and fish length, this study also found a potential correlation between water velocity and water depth. This present study's data, however, found no such correlation and thus it is inferred that, although correlated with water depth, occurrence of larger banded kokopu in deeper water is probably not due to the lower velocities encountered there in this aforementioned study.

Tree and tree fern roots, and undercut banks were all highly preferred cover types for banded kokopu in the streams studied, whose presences were all correlated to a significant degree (pers. obs.). Both trees and tree ferns provide greater bank support than pasture for undercutting to be sustained, and trees also provide for a higher more sheltered habitat encouraging native tree ferns (e.g. *Cyathea* and *Dicksonia* spp. etc.). The degree of bank-undercutting possible before the stream banks become unstable and slump is a direct consequence of both the degree and the type of riparian cover on the banks of the stream at that point. Trees, and tree ferns especially, have root masses and densities that form important components of the shear strength of soils offering greater protection against the gravity collapse of undercut banks in addition to armouring the soil against entrainment from floodflows (Hicks & Howard-Williams 1990; Collier *et al.* 1995a, 1995b). Like most pteridophytes, tree ferns form large adventitious root wads that frequently extend into the water contributing great protection from erosion. This is often seen when they are frequently found on the inside bends of stream meanders, due to their resistance to the momentum of the stream at that point (Plate 3.04/.). Indeed it has been proposed that tree ferns may be "keystone" species (Brendan Hicks, University of Waikato pers. comm.) through provision of more meandering and heterogenous stream morphologies than would be found without the presence of this plant type.

Plate 3.04/. The typical meandering nature of a small stream with the tree ferns ponga and mamaku (*Cyathea dealbata* and *C. medullaris* respectively) primarily occurring on the inside margins of bends. Sawmill Stream, Coromandel.

Although coarse instream woody debris has been noted as a preferred cover type by many authors (Stokell 1955; Graham 1974; Hopkins 1979b; McDowall 1975; 1990; Main, 1988) this study failed to find it as important as the other commonly encountered cover types. The proportion of fishes found within the arbitrary 0.6 m of no cover was also surprisingly high, and this unintuitive result thus brings this subjective distance into question. Main (1988) found woody debris a much more highly utilised habitat type than this study (a preference of almost 1.0) and this may simply be due to his use of a greater encompassing distance when determining items of used cover (1.0 m to this studies 0.6 m). A more accurate approach to answering this question of cover preferability is still a desirable research goal, and could possibly source the distance value to be used for determining cover types used from behavioural studies such as aforementioned remote video monitoring (Chapter 1 & 2).

Finally, whilst for a nocturnal species data collected at night may seem more relevant to a species' habitat requirements than habitat preference models generated from data collect during the day, these latter models will also be of value in both understanding the ecology and assisting with the conservation management of this species. It is clearly a flaw in logic to expect that a species that occupies a specific narrow habitat type at night will necessarily occupy the same habitat in the day when it is demonstrating a different activity and behaviour. Furthermore, the legitimate transfer of habitat suitability criteria to waterbodies other than those on which they are gathered clearly depends on habitat selection being consistent between those different bodies. It has been suggested that, for salmonids, velocity use is the most consistent of the habitat variables measured between rivers (Hayes & Jowett 1994). Given the number of streams and regions from which these habitat data were derived, and resemblance to results from similar studies, they should be applicable to most waterbodies in which banded kokopu occur.

Other important requirements of banded kokopu habitat

Other habitat variables besides those instream are noteworthy determinates of habitat suitability for many aquatic species. Indeed, the influence of a watershed on its stream's physical, chemical characteristics and consequent biology is well accepted. As Hynes (1975) stated so well "…in every respect the valley rules the stream…". Of these many catchment influences, the presence of riparian vegetation appears to be the most important out-of-stream habitat requirement for banded kokopu (Church *et al.* 1979; West, 1989; M^{c}Dowall, 1984; 1990; Wilding, 1996; 1997; Hicks & M^{c}Caughan, 1997). Immediate effects of the streamside vegetation also appear to play a very important part with many other native galaxiid species such as the shortjawed kokopu *Galaxias postvectis* (West, 1989; McDowall *et al.*, 1996), and mudfish *Neochanna* spp. (McDowall, 1990; Hicks & Barrier, 1996). These fishes have also been found to show strong associations with the marginal vegetation of their waterway environs.

With their typical stream temperatures being low (Taylor & Main, 1987), it is no surprise that banded kokopu have one of the lowest lethal temperature tolerances (Crawshaw, 1977) of the 12 native species tested so far (Simons, 1986; Richardson *et al.*, 1994). Temperatures found in deforested streams (Williamson, 1986; Quinn, 1992; Collier *et al.*, 1995a, 1995b; Wilding, 1997) are inclined to being significantly greater than those in streams with shading riparian vegetation Banded kokopu are also sensitive to low levels of dissolved oxygen (Dean & Richardson 1997) further decreased in warmer water (more commonly encountered in deforested streams) which will additionally compound their avoidance of habitats in these modified environments.

Adults of banded kokopu are often found in waters of quite low pH (Taylor & Main 1987; Collier *et al.* 1990; McDowall, 1990) and experimental data has indicated that banded kokopu appear to have a low pH preference (McDowall & Eldon, 1980; Richardson & Barrier, 1995; West *et al.*, 1997). It should also be noted that pH values may also synergistically affect the toxicity of other pollutants present, especially ammonia (Richardson, 1997).

Increased suspended sediments from erosion (Dons, 1987) may express itself both by inhibition of recruitment (Boubée *et al.*, 1997), and by impediment of their visual feeding ability (prey identification and capture) (Rowe & Dean, 1998). Increased wind across the surface of the water heightened by a lack of sheltering vegetation will create a “white-noise” or irregular background disturbance further impeding nocturnal detection of prey at the water’s surface. Feeding at the surface in habitats with less wind present as a consequence of significant riparian vegetation would facilitate readier prey detection and capture.

A stream’s riparian characteristics can greatly influence the nature and amount of allochthonous terrestrial invertebrate input (Collier *et al.* 1995a, 1995b), and (as previously noted) banded kokopu are widely regarded as feeding predominantly opportunistically on allochthonous terrestrial invertebrate input (Main & Lyon, 1987; Main, 1988; West, 1989; McDowall, 1990; Halstead 1994; Hicks, 1997). It may be concluded that a loss of riparian vegetation will typically result in a gross reduction in the biomass of these terrestrial invertebrate prey (Cadwallader *et al.*, 1980; Edwards, 1995), as well as important aquatic invertebrates (Collier, 1995a). Loss of riparian cover may also increase the predation risk from such noted avian piscivores as kingfishers (*Halcyon sanctus*) and whitefaced herons (*Ardea novaehollandiae*) (Church *et al.*, 1979; McDowall, 1990) which are more common in open areas (Reader’s Digest, 1985). Other predation (Hayes *et al.*, 1989) and further competition that may also affect banded kokopu distribution in some instances, are encountered from introduced salmonids (especially brown trout *Salmo trutta*) (see Townsend & Crowl (1991) for a list of contributing authors to this subject).

Stable coarse woody debris has been known to divert and obstruct water flow in such a way as to create a diverse range of pools resulting in an increased level of structural diversity that would not exist without this debris being present (Evans *et al.*, 1993). No doubt this stream modifying phenomenon is of especial importance in New Zealand where woody material is typically much less common than in comparable overseas lotic systems (Winterbourn *et al.*, 1981). Furthermore deforested streams tend to be significantly narrower, to around half their width in native forest (Davies-Colley, 1997) leading to half or less of the available stream channel habitat overall.

I propose that, whilst some consequences of the presence of significant riparian vegetation may play a more significant role e.g. increased terrestrial invertebrate input and improved stream morphology; other effects, however, such as decreased water temperatures (whilst a weaker influence) are not inconsequential as they yet further add to the value of these habitats (summarised in Table 3.05/.).

Table 3.05/. Summary of the effects of an absence of a stream's riparian known to influence banded kokopu abundance in stream, and the significance of these effects.

Typical differences of a forested stream to one without a significant riparian	Significance of these differences
Lower and more stable water temperatures	Banded kokopu have low water temperature tolerances
Decreased levels of incident light illuminating stream to a higher degree	Reduced competitive advantage over introduced visual feeding salmonids
Increased photosynthetic inhibition by shade	Periphyton blooms discouraged
More instream woody debris	More instream cover and pool habitat; greater degree of flood protection
Greater degree of bank undercutting	More avian predator protection
Increased ammonia if new landuse is agriculture	Highly toxic to most freshwater life
Increased invertebrate food abundance in stream	Important prey items in kokopu diets
Decreased and less variable pH	Kokopu prefer lower pH waters
Decreased suspended sediment levels	Inhibition of whitebait migrations and diurnal detection of prey
Altered stream morphology	Less pools, reduced water surface area
Greater water surface disturbance by wind	Background "white noise" inhibits nocturnal detection of prey
Interaction and summation of above differences ⇒	Compromised banded kokopu habitat, consequent decreased abundance

CHAPTER 4.

Application of Banded Kokopu Habitat Preferences to an IFIM Situation.

Habitat modelling on the basis of just a few basic habitat preference parameters provides…

> *"...fundamental data for more complex flow assessments on the basis of habitat to provide a rational and defensible approach to the allocation of water by taking into consideration the hydraulic response of the river to changes of flow and the potential effect on biota."*

Jowett (1996a)

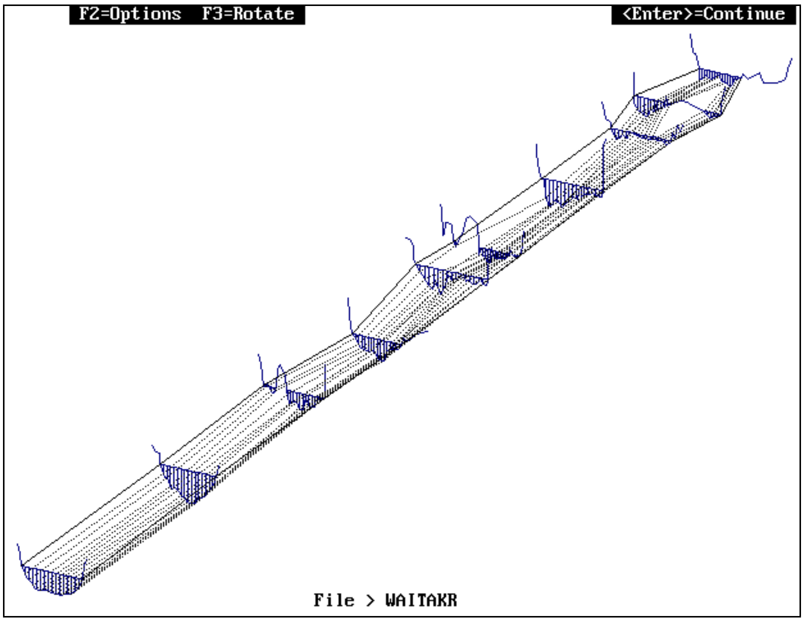

Figure 4.01/. Three-dimensional isometric display of the reach of the Waitakere River surveyed.

4.0 Introduction

Many people subscribe to the school of thought that natural ecosystems are best preserved under their original natural, pristine conditions. Others may hold to a more management based approach that follows that given sufficient knowledge of a system and its linkages and processes, a higher biomass/ biodiversity etc. may be realised following a specific anthropic intervention. Whatever the paradigm one follows, it ensues that wherever people and nature must coexist that compromises must be made and satisfactory balances struck between competing uses. Because of this requirement for minimum instream flow recommendations possible given abstractions and other uses, various objective quantitative techniques have been developed over the last few decades. A number of these techniques are still simply based on historical flow levels with the simplistic rationale that factors such as habitat and water quality may be assumed to be satisfactory because the aquatic resident species have survived such conditions in the past. More complex biologically (rather than hydrologically) based, and holistic techniques such as the habitat methods e.g. PHABSIM (Physical HABitat SIMulation system) are now, however, also frequently used. The aim of these methods is explicitly to retain a suitable physical environment for the aquatic organisms that live in the stream (Jowett, 1996a).

One of the most common interferences to lotic water bodies is the abstraction of water for either a long duration e.g. irrigation, or even only for a short time e.g. peaking power during hydroelectricity generation. Indeed irrigation is a major pressure already on water volumes and flows in New Zealand, and their consequent ecology (McDowall, 1984), largely as a consequence of the dominance of pastoral land usage in low-lands (Taylor & Smith, 1997). Either interference has a potential to impact on the instream freshwater species populations in a negative manner. The ecological goal of most minimum flow methods is therefore to sustain existing instream life by recommending a minimum flow that is within the long-term tolerances of the species contained therein. In order to ascertain the effects of the changing flow (albeit maybe only for a short duration, or even infrequently) case-by-case modelling of the stream's morphology and relevant freshwater species' preferences for the basic habitat parameters of water depth and velocity, (i.e. a PHABSIM approach) is often considered the most reliable, most comprehensive, and most defensible objective methodology than assessments derived by other methods. In addition to these values, this is also the most widely used method in the United States (Jowett, 1996a).

This exercise involved a task for the Waitakere Golf Course (West Auckland) (Hicks & McCullough, 1998). The golf course were applying for resource consent (under the Resource Management Act (1991)) from the Auckland Regional Council (ARC) to irrigate their playing greens with water sourced from a specific site out of the nearby Waitakere River (Figure 4.02/.) (NZMS series 260 map Q11 coordinates: 26457.64787 for the exact point of abstraction).

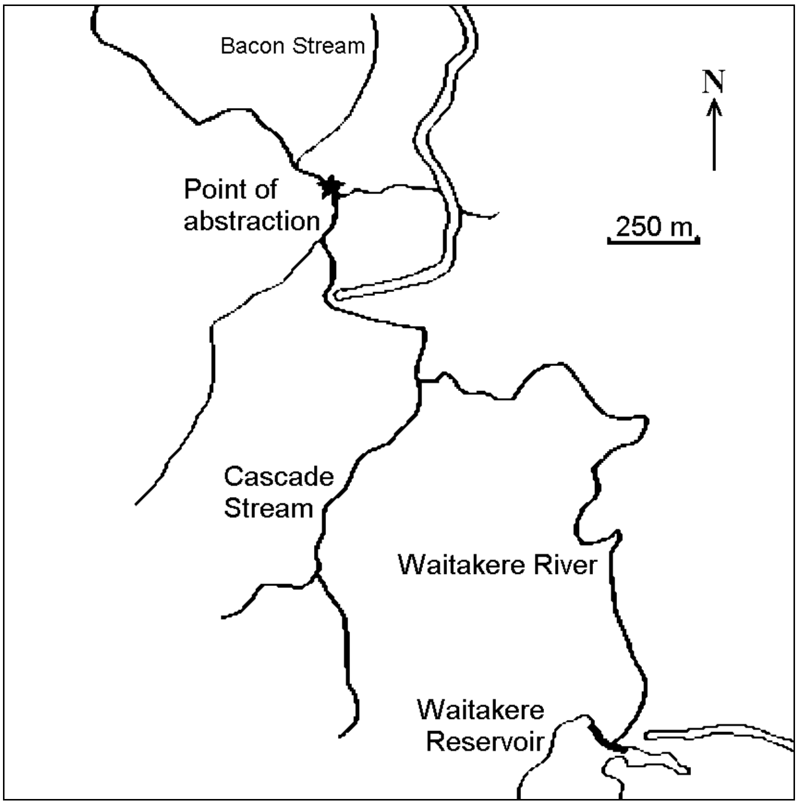

Figure 4.02/. Location of the Waitakere River and the proposed water abstraction site.

A small pool had already been constructed around the abstraction intake from which water had already been drawn from the stream for many years (Plate 4.01/.).

Plate 4.01/. Artificial pool at the point of water abstraction from the Waitakere River.

The 71 km^2 catchment of the Waitakere River has its headwaters, and more than half this catchment, in near-pristine cool-subtropical, kauri/ broadleaf-podocarp rainforest in the Waitakere Ranges (West Auckland). Some of the more common native riparian species found along the banks of the Waitakere River, from the proposed site of stream water abstraction downstream included swamp flax (*Phormium tenax*), rewa rewa (*Knightia excelsa*), tutu (*Coriaria arborea*), five finger (*Pseudopanax arboreus*) mahoe (*Melicytus ramiflorus*), manuka (*Leptospermum scoparium*), kowhai (*Sophora tetraptera*), and blechnum fern (*Blechnum capense*). Occasional spaces where a tall, woody riparian cover is absent were commonly completely dominated by the regional plant-pest montbrettia (*Crocosmia* x *crocosmiiflora*), which was no doubt also inhibiting native floral recruitment here. Surface soils are generally of a clay nature and give rise to a relatively high runoff (Waitakere Ranges Protection Society, 1978); geology is of Miocene andesite or andesitic sediments. Storm events are buffered by the Waitakere Water Supply Reservoir catchment upstream with a likely otherwise higher mean annual flow, and storm event frequency and intensity otherwise. This domestic water supply reservoir abstraction and impoundment clearly already impacts on the natural flow regime of the Waitakere River in what is an essentially negative manner. Very little historic, or even present day, hydrological data is, however, available on the effects of this construction on the river's natural flow regime (Alan McPike, Auckland Regional Council, pers. comm.).

The Waitakere River has had a long history of abstraction, being the location of the first major municipal water supply for early 20th century West Auckland. A timber dam was present on the river around the current dam's site as early as 1907, being later replaced by a 20 m concrete construction in 1910. This impoundment was further capped in 1928 by around 5 m to a total present day height of approximately 25 m. The reservoir's previous annual abstraction from this impounded body of water (which otherwise would have entered into the Waitakere River) for the last five years averages around 16, 000 m^3 a day, or about 0.185 $m^3/$ s. Overflow of the dam in winter months is understood to be relatively frequent, although summer lake levels of this magnitude are unlikely (Ray Turner, Water Care, pers. comm.).

The lower reaches of the river also flow through a highly ecologically significant wetland (Te Henga Wetland) (McCullough 1995; 1996a; 1998a) before finally mingling with the Tasman Sea approximately 11 km downstream from the study site at Bethells Beach. This wetland has formed from a mature dune lake originating about 6,000 years ago, when accumulating sand dunes blocked the Bethells Valley mouth and the path of the historic Waitakere River.

Approximately 0.0041 $m^3/$ s (maximum) of water was proposed to be drawn from this stream with a total abstraction volume of 119 $m^3/$ day (max). The river at this point has a one-year return period of 0.045 $m^3/$ s, a mean annual low flow of 0.0200 m^3/s, and a one-in-five-year low flow of 0.0134 $m^3/$ s (Marian Jenner, Auckland Regional Council, pers. comm). Technical information was therefore required on what the impact of this abstraction would be on the aquatic species of this stream.

Comparison with an historic flow method: the Tennant (Montana) method

This method is perhaps the most widely known of these simple methods, and is the second most popular in the United States where it was developed. In a concise description, it is simply based on the assumption that some percentage of the mean flow is required to maintain a healthy stream environment. When Tennant (1976) examined cross-section data from 11 streams situated in Montana, he found that whilst stream width, water velocity, and depth all increased rapidly from zero-flow to 10% of the mean flow to date; the rate of increase, however, substantially declined at flows greater than 10%. In addition, at and below 10% of this average flow; fish were crowded into the deeper pools, riffles became too small to facilitate access for larger fish, and increased water temperatures began to become limiting. Recommended minimum flows are therefore merely based on percentages of this average minimum flow to date, typically often further with differing percentages for the winter and summer months.

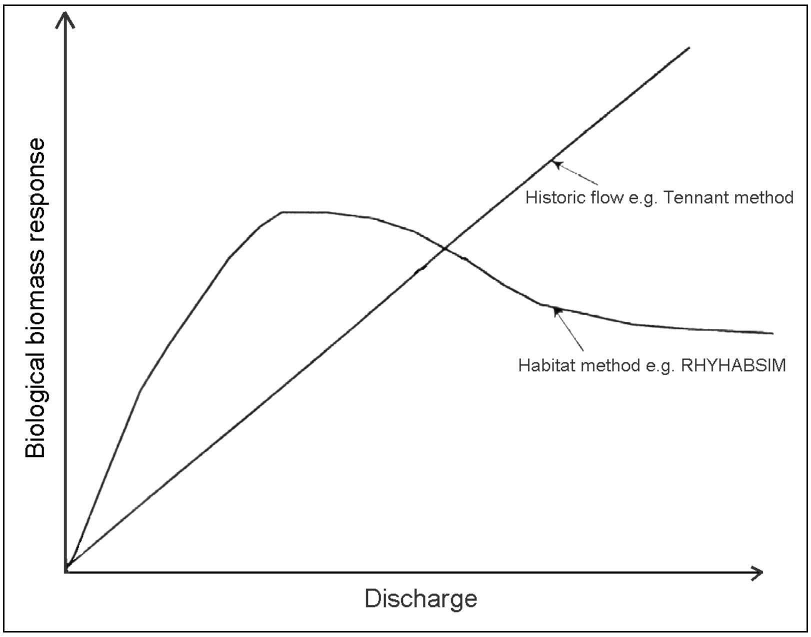

Figure 4.03/. Comparison of the typical relationship assumptions between flow and biological response (abundance) for the historic flow and habitat based methods in a hypothetical river. After Jowett (1996a).

It was decided to also investigate the minimum flow recommendations of this very different and simpler method to those recommendations of the more involved IFIM habitat model RHYHABSIM on this New Zealand river.

4.1 Methods

Field

Following a period of rain only days earlier in the area the Waitakere River was surveyed on 5 December 1997. Sampling (methodology after Jowett (1996b)) began with 200 m of the stream down from the proposed abstraction site being habitat mapped. The presence of a discharge input within this 200 m mapped reach (i.e. Bacon Stream) (Plate 4.02/.) which is generally considered undesirable due to its confounding nature on the response of the system to be modelled (Ian Jowett, National Institute of Water and Atmospheric Research, pers. comm.), was considered to be of little consequence as a result of its small contribution to the river, an likelihood of drying completely in summer. Ten cross-sections were then chosen from this 200 m mapped; 4 in pools, 3 in runs, and three in riffles. Data collected from cross-sections included a gauging with substrate type (classification after Jowett (1996b)) also being noted at each sampling point, and the size and placing of any immediate up- and downstream obstacles to flow e.g. boulders. Finally a profile of the stream cross-section both below and approximately 0.8 m above the water surface was derived by continuing the cross-section gauging up the banks either side to the desired elevation (Plate 4.02/.).

Plate 4.02/. Brendan Hicks executing a discharge gauging and cross-section profile of the Waitakere River next to the Bacon Stream confluence.

The stream was returned to 3 days later (8 December) to be sampled for the final time following no rain in the catchment. Sampling for this occasion simply consisted of noting the new stage heights at each cross-section following the stream's consequent lowering water level, and a single gauging at the most appropriate site (the most even run).

Data Analysis

Resulting stream morphological and gauging data was analysed on RHYHABSIM (Jowett 1996c); a PHABSIM program for DOS; on a PC-computer. Modelling occurred with substrate set to on, and velocity-distribution factors (VDFs) also on. VDF's were further manually adjusted to more accurately represent the character of the 200 m reach.

A National Freshwater Fish Database search (McDowall & Richardson, 1983; Richardson, 1989) was also executed for the Waitakere River Catchment, and this information on the knowledge of species' presences for given dates was combined with anecdotal and personal observations to produce a table of species whose presence had been noted downstream from the proposed water abstraction site. Fish species modelled were those noted to be historically found in the catchment (National Freshwater Fisheries Database, 1998) less those not found in the previous 20 years (Table 4.01/.). An updated file of the relevant freshwater fish species to be modelled, including banded kokopu habitat preferences from the research discussed in Chapter 3, were then used for the RHYHABSIM fisheries analysis.

Table 4.01/. Freshwater fish species noted in the Waitakere River mainstem.

Common name	Scientific name	Sites noted (date of latest recording)
Shortfinned eel	*Anguilla australis*	1 (7/95), 4 (9/96), 7 (11/93)*, 8 (2/89)
Longfinned eel	*A. dieffenbachii*	4 (9/96), 7 (11/93)*
Banded kokopu	*Galaxias fasciatus*	7 (11/93)*
Inanga	*G. maculatus*	1 (7/95), 7 (3/65)
Crans bully	*Gobiomorphus basalis*	7, (12 /97)*
Common bully	*G. cotidianus*	1, (7/95), 4 (9/96), 7 (11/93)*, 8 (2/89)
Redfinned bully	*G. huttoni*	4 (2/89), 7 (11/93)*, 8 (2/89)
Common smelt	*Retropinna retropinna*	1 (3/65)

* Indicates a personal or anecdotal observation, or Woodard (1994). All other observations are from the New Zealand National Freshwater Fish Database (National Institute of Water and Atmospheric Research, Hamilton).

Aquatic macroinvertebrates modelled were those for which habitat preference data was both available for (Jowett & Richardson, 1990), and which had also been noted in the Waitakere River immediately downstream of the proposed abstraction site (Towns, 1979). This allowed modelling of 5 of the 13 species that had been noted at sites in the river close to the point of abstraction (Table 4.02/.). Food producing waters, a suitability function simply relating benthic aquatic macroinvertebrate biomass to the three main habitat variables was also modelled, with the significance of these organisms being that they form a significant diet component of many freshwater fish species.

Table 4.02/. Aquatic macroinvertebrate species noted in the Waitakere River and modelled.

Family	Species
Leptophlebiidae	*Deleatidium* spp.
Siphlonuridae	*Coloburiscus humeralis*
Hydropsychidae	*Aoteapsyche* spp.
Sericostomatidae	*Olinga feredayi*
	Pycnocentrodes spp.

The initial flow range modelled was from 0.0200 m^3/s to 0.0093 m^3/s, representing the range of mean annual low flow periods in summer with no abstraction, to one-in-five-year low flows with the proposed rate of abstraction. Given the potentially significant effects of the abstraction of the upstream dam, a further range from 0.100 to 0.040 m^3/s was also modelled. This range was intended to represent a likely mean annual low flow at this point if the dam was not present. The value of 0.100 m^3/s was calculated from the premise that the expected flow from the dam based on current rates of abstraction will decrease in the same proportion as does the flows further down the river during dry periods. The one-year return period of 0.045 m^3/s is therefore expected to reduce to a mean annual low flow of 0.020 m^3/s; a reduction factor of around 44%. Thus 44% of the expected flow given the dam's absence of 0.185 m^3/s gives an estimated mean annual low flow without the dam of 0.080 m^3/s. A further addition of 0.020 m^3/s (flow contributed by catchment further downstream) gives a total expected mean annual low flow at the proposed point of abstraction of 0.100 m^3/ s.

Comparison with the Tennant (Montana) method

Total WUA for all species including native fish (all sizes of banded kokopu only) and invertebrates were modelled for mean annual flows expected given the dams absence. Flows modelled were those for the Tennant values of 200% to < 10% of mean annual flows.

4.2 Results

The average discharge across all ten cross-sections at the study site following rain on the first survey trip was 0.147 m^3/s. Following a dry spell over the next few days this flow eased to a second gauging of 0.0692 m^3/s.

The channel of the Waitakere River is rather rectangular in cross-section, notably different in shape from the more typical parabolic nature of most New Zealand waterways (Fig. 4.04/.). By far the greatest habitat type present were pools of velocities from a high of 0.45 to 0 m/s and depths to 0.5 m. This habitat type accounted for 78.6% of the habitat noted in the mapping of the 200 m reach downstream from the abstraction point. The two other main habitat types noted, run and riffle, represented only 17.0% and 4.5% respectively of the remaining habitat.

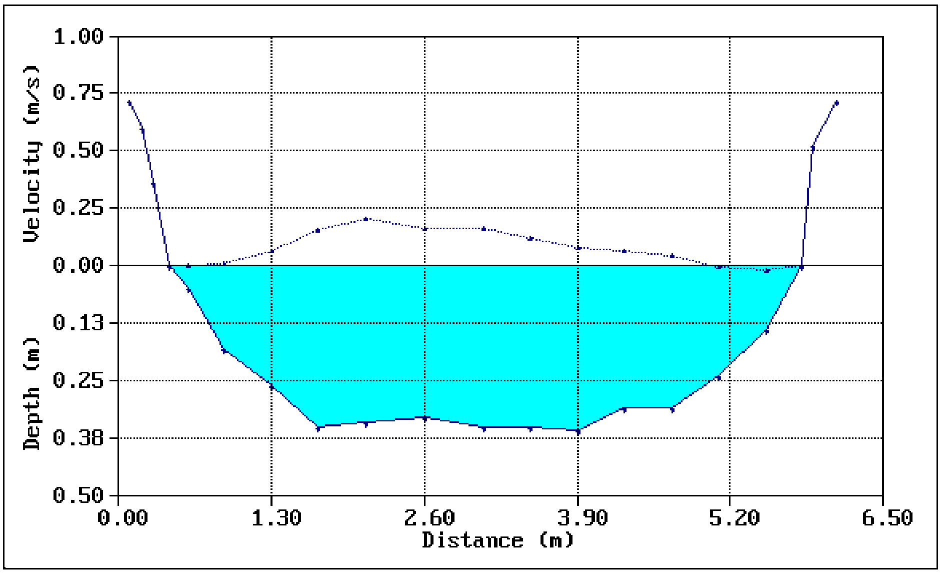

Figure 4.04/. Cross-section of a highly channelled pool (pool 10) in the Waitakere River.

As can be seen from Fig. 4.05/., the site of the proposed water abstraction location is immediately above a steep increase in river bed altitude and hence channel gradient. It is likely that this increased incline represents itself as either cascades or rapids; either of which would form a significant obstacle to poorly climbing diadromous species trying to enter this site via a migration from the sea. For this additional reason also, the habitat preferences of both the poorly climbing species of inanga and smelt were not considered in the RHYHABSIM assessment. It was felt that their sighting in the immediately affected region of the proposed stream water abstraction was of insufficient weight given in addition to their rarity of records and the duration of these records since their last sightings (McDowall, 1993).

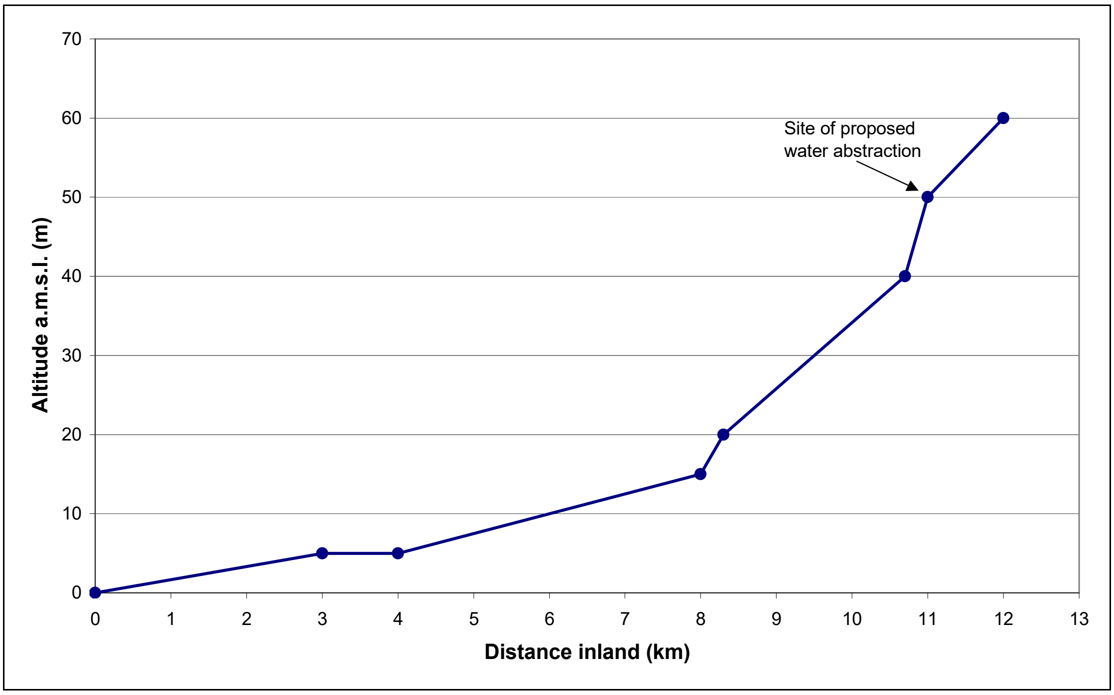

Figure 4.05/. Inland distance/ altitude topographical analysis of the Waitakere River.

Effect of abstraction on instream habitat

The effect of the dam on the expected habitat flow regime without the resultant flow reduction caused by its significant abstraction, is with average depths across the reach only falling 19% from 0.21 m to 0.17 m, but with average water velocities decreasing 60% from 0.10 m/s to 0.04 m/s. The proportion of the reach as pools increased from 86% to 95% of the reach. As a result riffle habitat was also lost more readily than the deeper, but slower, run habitat. Abstraction by the Waitakere golf course at the proposed rate of only 0.0041 m^3/s from a historical pre-dam mean annual low flow of around 0.100 m^3/s would therefore have been inconsequential to the flow and habitat regime encountered in this larger and less precarious water body.

Reduction of discharge from a mean annual low flow of 0.020 m^3/s to 0.016 m^3/s following abstraction resulted in an average loss of only 0.01 m depth and 0.01 m/s average water velocity along the surveyed reach. No notable changes in habitat structure from the roughly 95% dominated pool regime at 0.0200 m^3/s was also observed. Wetted perimeter changed over these flows from 3.90 m to 3.82 m. Flow reduction following proposed levels of abstraction during a one-in-five-year low flow of 0.0134 m^3/s would, however, result in a flow of only 0.0093 m^3/s in this reach. This would further result in average depth losses from 0.17 m to 0.16 m of only 0.01 m, but average water velocity decreases from 0.04 m/s to 0.02 m/s; a weighty 50% reduction. Once again the pool dominated regime remains largely unchanged at around 95%. The nature of the reach was therefore not greatly changed absolutely, although the mean velocity decrease represents a considerable relative decrease (50%). The wetted perimeter only decreased slightly (3.77 m – 3.64 m) over this flow decrease.

Effect of abstraction on resident native freshwater fishes

Flow reductions within the range modelled for all species of fish other than banded kokopu for the effects of the proposed abstraction during a year of average low flows (0.0200 – 0.0159 m^3/ s) were followed by notable reductions in WUA (Fig. 4.06/.).

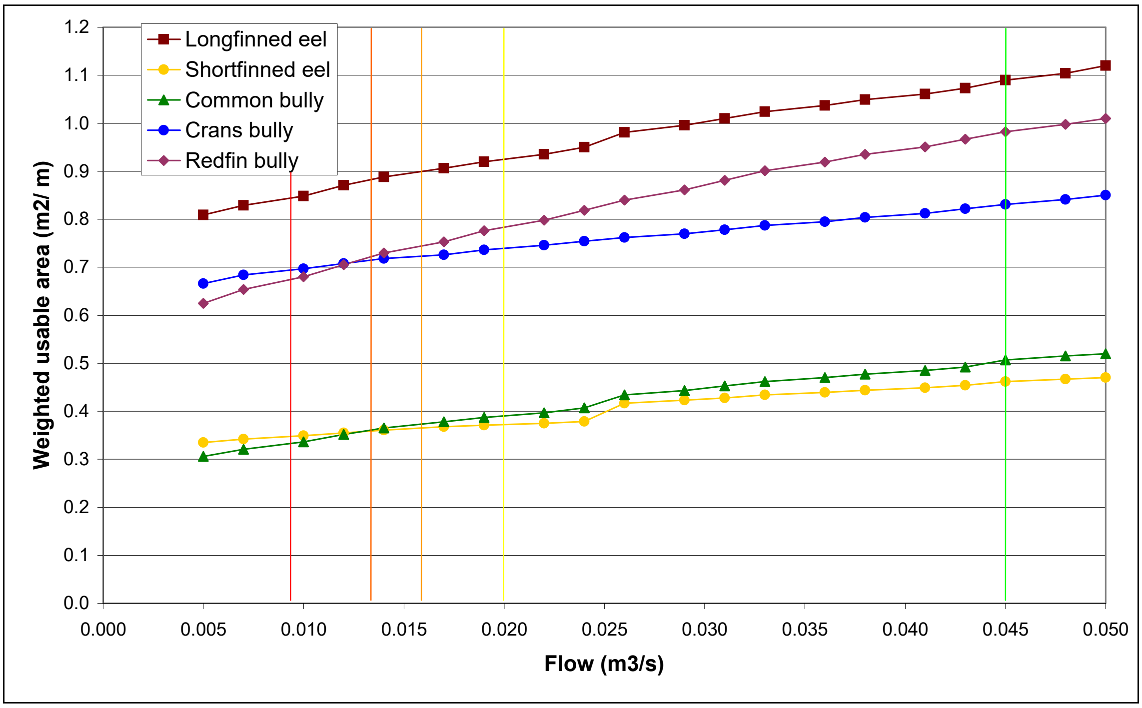

Figure 4.06/. Effect of flow reductions on weighted useable area (WUA) for freshwater fishes noted in this section of the Waitakere River. See text for details.

Green vertical lines indicate one year return period flows with no abstraction, yellow lines indicate mean annual low flows without abstraction, light orange lines indicate mean annual low flows with abstraction, dark orange indicate one-in-five-year low flows without abstraction, and red lines indicate one-in-five-year low flows with abstraction. Shortfinned eels showed the least decrease in WUA and redfinned bully and longfinned eel the most. All changes were minor given that most of their habitat available has previously been lost following construction of the reservoir.

Further reductions in flow to 0.0134 m^3/s would be expected on an average of every five years at this point, with proposed abstraction lowering the flow to 0.0093 m^3/s. This decreased in flow could be expected to result in the following changes in weighted useable for these freshwater fishes observed in this section of the river (Table 4.03/.);

Table 4.03/. Predicted change in weighted useable area for fish species less banded kokopu for a 0.0041 m^3/s abstraction during a one-in-five-year low flow (0.0134 m^3/s).

Fish species	Change in WUA	% change in WUA
Longfinned eel	0.88 – 0.85	- 3.4%
Shortfinned eel	0.36 – 0.35	- 2.8%
Common bully	0.36 – 0.33	- 8.3%
Crans bully	0.71 – 0.69	- 2.8%
Redfinned bully	0.72 – 0.68	- 13.9%
Total	3.03 – 2.93	- 3.3%

There appears to already be little available habitat for banded kokopu in the reach studied during any magnitude of flow (Fig. 4.07/.). The WUA of 1+ fish slightly decreases in response to increased flows, and 0+ fish WUA slightly increase. No notable changes appear to be observed for all sizes of banded kokopu when observed together. Changes in WUA during one-in-five-year low flows are therefore likely to be observed as a slight decrease in weighted useable area for 0+ fishes only, with little changes for 1+ fishes, or for all sizes in general (Table 4.04/.).

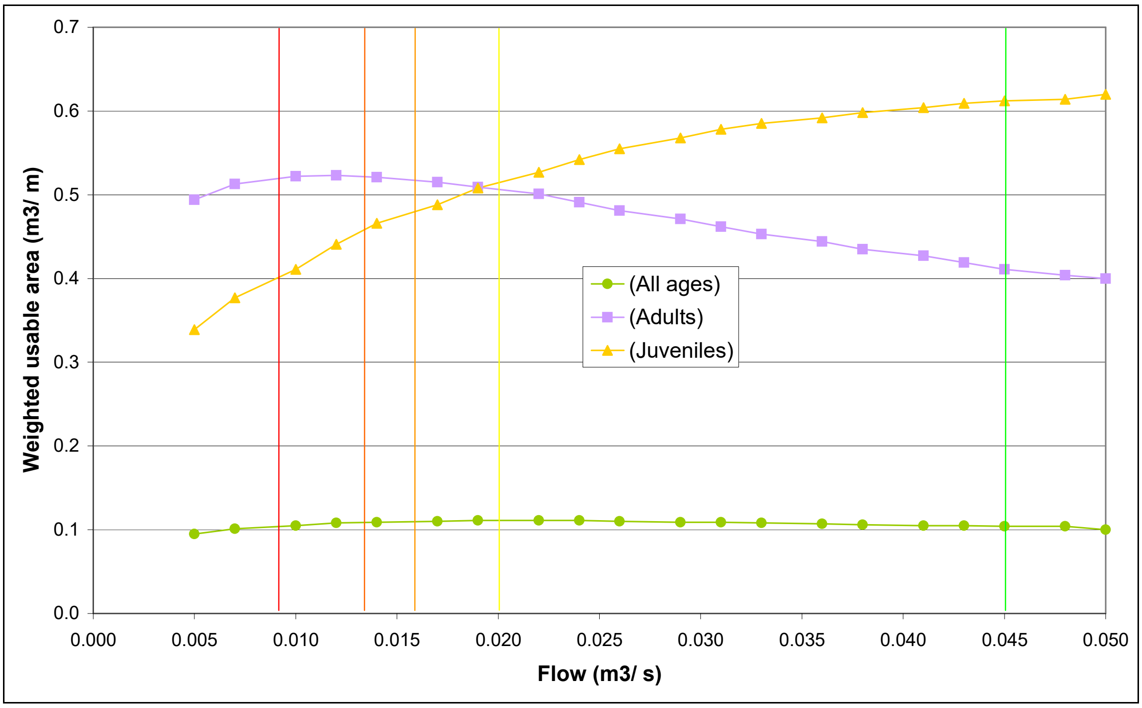

Figure 4.07/. Effect of flow reductions on weighted useable area (WUA) for both all sizes, and 1+ fishes and 0+ fishes separately, of banded kokopu present. See text for details.

Table 4.04/. Predicted change in weighted useable area for different banded kokopu size classes for a 0.0041 m^3/s abstraction during a one-in-five-year low flow (0.0134 m^3/s).

Banded kokopu age	Change in WUA	% change in WUA
0+ fishes	0.46 – 0.41	- 10.9%
1+ fishes	0.11 – 0.11	0%
All sizes	0.53 – 0.53	0%
Total	1.10 – 1.05	- 4.5%

Effect of abstraction on aquatic macroinvertebrates

Slight decreases of WUA were observed for all invertebrate species modelled following abstraction from a mean annual low flow, and also for the model of food producing waters (Fig. 4.08/.). Relatively greater decreases were noted for species with the most WUA than for those with minor values.

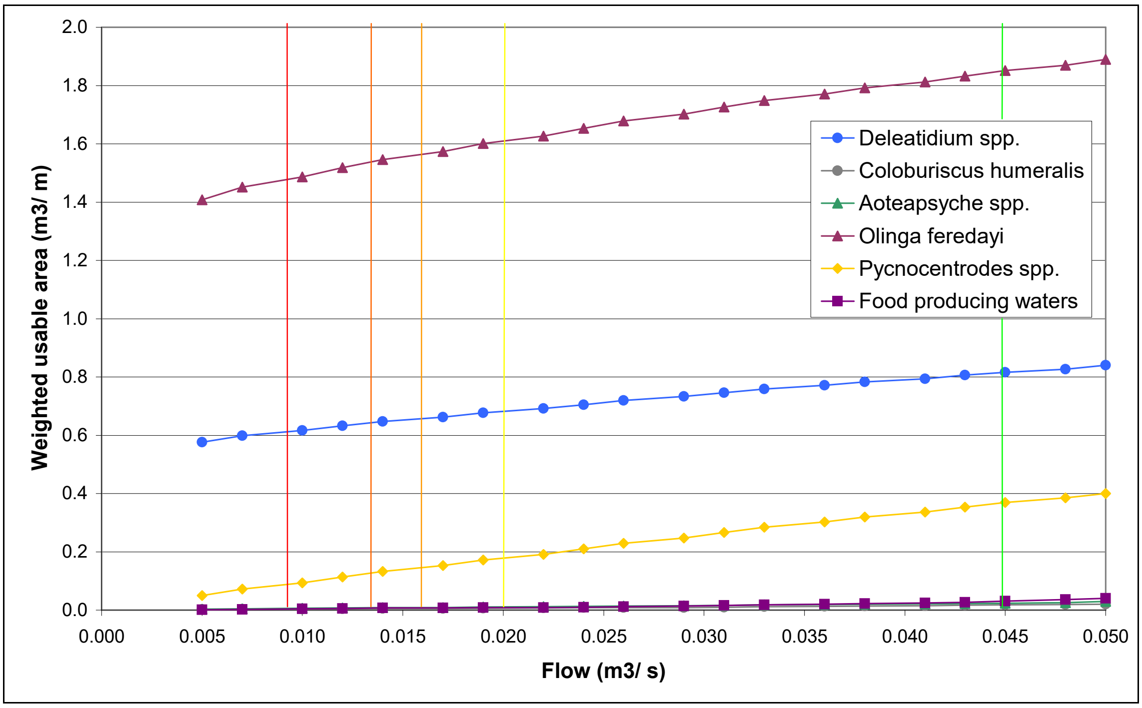

Figure 4.08/. Effect of flow reductions on weighted useable area (WUA) for aquatic invertebrates found in nearby sites in the Waitakere River. See text for details.

Further distinct decreases in WUA following proposed rates of in a one-in-five-year low flow abstraction were only noted for the cased caddisfly *Olinga feredayi*. WUA for all other individual species and for food producing waters was already too low to accurately detect any of the slight decreases. Even paltry changes, however, may be relatively considerable given the already extremely low values for the WUA of these species.

Table 4.05/. Predicted change in weighted useable area for aquatic macroinvertebrates for a 0.0041 m^3/s abstraction during a one-in-five-year low flow (0.0134 m^3/s).

Species	Change in WUA	% change in WUA
Food producing waters	0.0 – 0.0	0%
Deleatidium spp.	0.6 – 0.6	0%
Coloburiscus humeralis	0.0 – 0.0	0%
Aoteapsyche spp.	0.0 – 0.0	0%
Olinga feredayi	1.6 – 1.5	- 6.3%
Pycnocentrodes spp.	0.1 – 0.1	0%
Total	2.3 – 2.2	- 4.3%

Tennant (Montana) method/ Habitat method comparison

Although extrapolation outside of the surveyed flows was required to determine total WUA for all instream species modelled, this was not considered a significant problem due to the wide channel of the river depths remaining within the above water bank measurements.

Table 4.06/. Comparison of instream flow regimes and their predicted effects on stream life given by the Tennant method and by RHYHABSIM of all modelled species Weighted Useable Area predictions for the seasons of spring and summer. From Tennant (1976).

Base flows (% mean)	**Discharge (m^3s^{-1})**	**Instream Flow Method Predictions**	
		Tennant	**RHYHABSIM – WUA (m^2/ m)**
< 10%	< 0.0023	severe degradation	< 5.961
10%	0.0230	poor/ minimum	5.961
30%	0.0690	fair/ degrading	7.126
40%	0.0920	good	8.086
50%	0.1150	excellent	9.430
60%	0.1380	outstanding	9.927
60-100%	0.1380-0.2300	optimum range	9.927 - 11.499
200%	0.4600	flushing/ maximum	12.820

Expected total WUA for the species modelled in this study prior to the abstraction of the Waitakere River is estimated at around 20.5 m^2/ m for one-year return period flows. Following this abstraction this previous value of WUA is thought to have decreased by more than a 40% reduction in WUA to a present day 12.0 m^2/ m (Fig. 4.09/.).

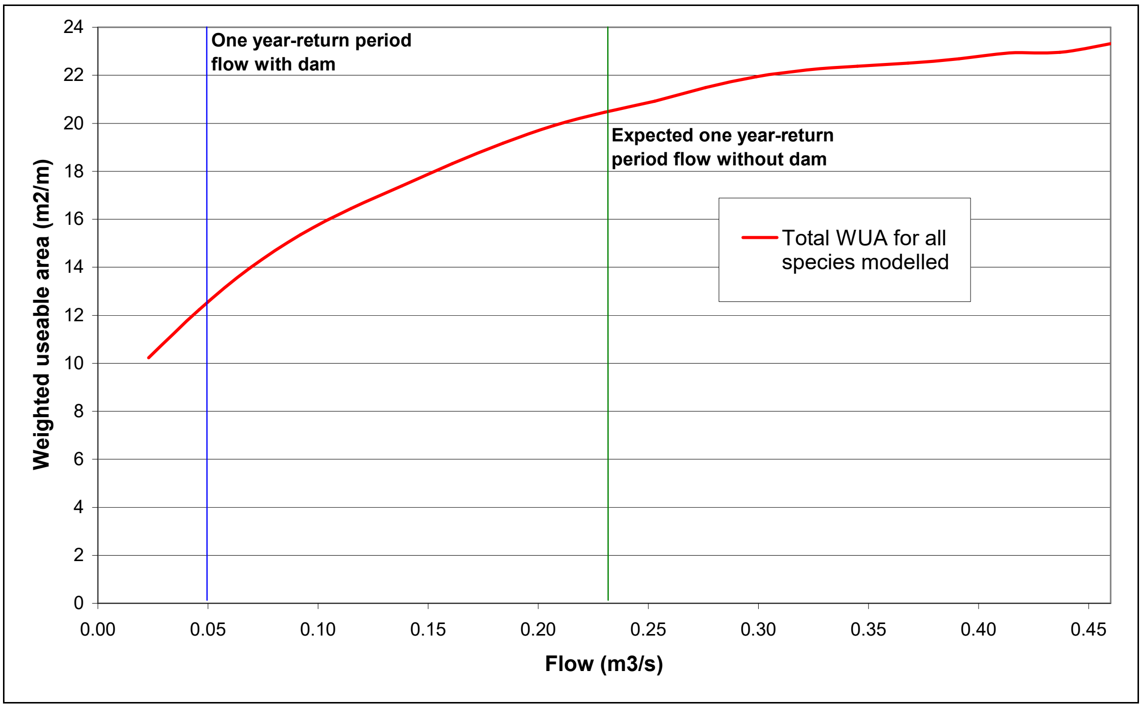

Figure 4.09/. Relationship of total WUA for all aquatic organisms modelled for changing flow.

4.3 Discussion and Conclusions

Effect of abstractions on the Waitakere River

The presence of the significant abstraction by the Waitakere Dam upstream of the study site provides for notably important confounding effects for both the RHYHABSIM and the Tennant methods.

Expected flows through the reach surveyed in absence of the Waitakere Dam would be expected to be approximately five times greater in all seasons, given the current difference between realised one-year return period of 0.045 m^3/s flows at this point and these current flows plus the Waitakere Reservoir's mean annual abstraction of 0.185 m^3/s (a total of 0.230 m^3/s). The presence of this upstream dam and consequent abstraction have largely resulted in the Waitakere "River" at this point being rather little more than an actual stream in magnitude.

The proposed abstraction of 0.0041 m^3/s may therefore be seen to represent only around 2% of the total mean current abstraction from this river including the reservoir, or around 4.7% when irrigation is occurring in summer during typical mean annual low flows. It would appear that current abstraction by the reservoir has already resulted in low flows which have placed all of the species modelled below points of inflection for declining WUA. For most species in this river this accelerated rate of decreasing WUA had occurred between the flows of 0.07 and 0.05 m^3/s. No inflections of WUA were observed within the flow ranges produced by the proposed abstraction during either of the mean annual low flow of 0.0200 m^3/s, or the one-in-five-year low flow of 0.0134 m^3/s. It would therefore appear that previous abstraction by the reservoir further upstream has already resulted in low flows which have placed all of the species modelled below significant points of inflection for declining WUA. For most species in this river this accelerated rate of decreasing WUA had occurred between the flows of 0.07 and 0.05 m^3/s. No significant inflections were observed within the flow ranges produced by the proposed abstraction during modelling of either the mean annual low flow of 0.0200 m^3/s, or the one-in-five-year low flow of 0.0134 m^3/s.

Changing suitability of hydraulic variables are unlikely to be as important as these wetted perimeter changes as depth and velocity changes are very slight. Most of the decreases in WUA following decreased flow are therefore probably mainly due to consequent decreases in wetted area. These changes are, however, modest as a consequence of the deep and rectangular nature of the stream bed (Fig. 4.04/.) providing for little over-topping of banks following an increase in flow. This channel shape and capacity is probably due to the considerably greater magnitude of flows historically present in this system.

All fishes modelled showed decreases of approximately 5% of their previous WUA values following abstraction from either mean annual or one-in-five year low flows. Whilst some are of greater conservation value both regionally and nationally (redfinned and crans bully) none of these fishes have special conservation considerations (McDowall, 1990). These species also all have additional suitable habitat in many of the tributaries of the Waitakere River itself (McDowall, 1990; National Freshwater Fish Database, 1998).

Although banded kokopu are of significant conservation value (Swales, 1991; Swales & West, 1991) the Waitakere River, even without the Waitakere Golf Course abstraction, already appears to have little available habitat for this species in this reach

(Fig. 4.07/.). This largely appears to be through the low preferences given to the nature of the predominantly substrate found here (primarily cobbles) (Fig.'s 3.05/., 3.10/.) and shallow water depth during typical summer low flows (Fig.'s 3.04/., 3.09/.). Changes in habitat are likely to be observed as a decrease in weighted useable area for 0+ fish only, with little changes for 1+ fishes, or for all sizes in general. There is, however, also significant quality habitat and populations of this species in the tributaries of the river, and indeed in much the region itself (McDowall, 1990; National Freshwater Fish Database, 1998). In light of the presence of these alternative habitats the expected result of a the slight decrease in weighted useable habitat for 0+ banded kokopu following abstraction in summer is likely to be of relative inconsequence.

A reasonable objective for sustainable management is that minimum flows should ensure that the level of protection for aquatic species is adequate, rather than attempting to ensure that populations remain unchanged (Jowett, 1996a). However, because weighted useable area is only a measure of *potential* habitat availability, other factors that clearly control the *actual realised* habitat usage (as noted with intraspecific predation in Chapter 2, e.g. maintenance of food supply, migration barriers etc.) must be also included in a complete analysis of the instream flow requirements of the waterbody's biota (Mosley & Jowett, 1985).

It has therefore been concluded (Mosley & Jowett, 1985) that, in order of importance, the following environmental variables control the biomass of a given fish species in any given watercourse, will the physical factors appearing to be more important. Therefore in decreasing order of prevailing significance; depth, velocity, temperature, food supply, riparian cover, and competition can be seen to be the more important environmental factors driving a fish species' abundance at a given location (Fig. 4.10/.). Other factors such as predation, substrate, dissolved oxygen etc. can be considered of a lesser importance, partially as their natures are often a consequence of the magnitude of the aforementioned more significant variables e.g. water temperature will strongly influence dissolved oxygen levels. In consequence of these facts, it is more logical to determine flow requirements that are based on hydraulic conditions that meet specific biological requirements, rather than merely the hydraulic parameters as an end in themselves (Jowett, 1996a). Because of this the physical habitat component

(PHABSIM) of the IFIM process is very often the most frequently used component, often to the complete exclusion of the other component models (Orth, 1987).

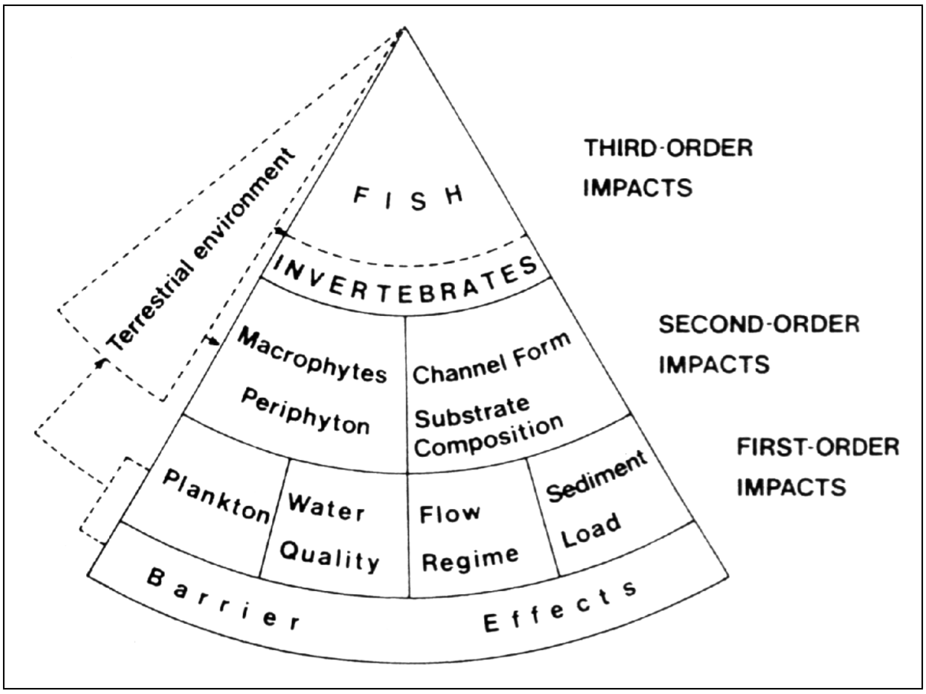

Figure 4.10/. A hierarchal framework of the impacts of a river impoundment, the model is also applicable to water abstraction situations. From Petts (1984).

Whilst this study centred around preservation of the native freshwater fisheries of the river, it is therefore also important that invertebrates are also considered in instream flow assessments because of their fundamental importance in stream ecosystems (Orth, 1987). For example the unsampled benthic invertebrate larvae will clearly form a major component of the crans and redfinned bullies found in this stream (McDowall, 1965; 1990), amongst other fish species. So whilst benthic invertebrate density does not appear to be related to native freshwater fish distribution (Jowett, 1996a), protection of a species' food source, and factors other than those evaluated by the sole application of the PHABSIM model, are imperative to their continued survival in that system. This factor therefore needs to be considered before recommendations are made. This is especially so when the habitat preferences of the primary species concerned are broad, as typically are the habitat requirements of many native freshwater fishes e.g. native eels and common bully. Many species of mayfly (order Ephemeroptera), stonefly (order Plecoptera), and caddisfly (order Trichoptera), however, are considered sensitive to declines in habitat quality (Collier, 1995a).

Abstraction, nonetheless, failed to produce significant changes in most of the invertebrates modelled, no doubt due to the already unsuitable nature of the slow water pool dominated reach. This is seen with preferences for water velocity being high (typically < 0.5 m/s) for all species showing decreasing WUA following decreased flows (Jowett & Richardson, 1990). The only two species which showed high WUA in this reach (*Olinga feredayi* and *Deleatidium* spp.) were therefore those tolerant of the typically low water velocities encountered over much of this reach in the pools.

Both *Aoteapsyche* and *Coloburiscus humeralis* have been observed in almost all of 14 samples taken around this of this part of the river previously (Towns, 1979) yet the RHYHABSIM model predicted very low densities at all flows. This is due to the low proportion of the riffle habitat that mapping found to be present in the entire 200 m reach which RHYHABSIM is modelling, whilst actual abundances within these riffles may be quite high (Brendan Hicks, University of Waikato, pers. comm.), rather than an inaccuracy in the model itself. One way of testing for changes within these riffle habitats would be to program riffles in RHYHABSIM to represent the whole reach i.e. allowing the three riffle cross-sections measured to add to the 100% of habitat type present in the 200 m reach.

Although not modelled, temperatures and oxygen saturations may reach undesirable levels for both fishes (Simons, 1986; Richardson *et al*., 1994) and invertebrates (Quinn *et al*., 1994) during low flows in summer months. The former situation due to the more rapid heating of smaller lotic bodies (Collier *et al*., 1995a, 1995b) the latter due to the decreased ability of this now warmer water to absorb oxygen. Excessive periphyton may be encouraged by these warmer water temperatures (Collier, 1995b), and will further contribute to increased oxygen and pH variability through respiration overnight following photosynthetic activity in the day (Biggs & Price, 1987). Furthermore, a synergistic build-up of silt may additionally occur when the algal growth on a stream bed flocculates fine suspended particles (Church *et al*., 1979).

In conclusion, the predictions of the habitat method of estimated WUA to measured WUA has been found to be fairly reliable in New Zealand (Mosley & Jowett, 1985). The accuracy of the IFIM (PHABSIM component) approach to evaluating the effect of these changing hydraulic conditions on the post-flow change biomass of aquatic organisms has, however, received criticisms both internationally and from within New Zealand's scientific community (e.g. see Gore & Nestler (1988) for a review of the criticisms put forward by a number of authors in the international literature, and Scott & Shirvell (1987) for a review of New Zealand experiences). A primary concern arising from these discourses relates to these models frequent inability to accurately predict realised species biomass; with an estimation of the weighted useable area (WUA) often being utilised instead as an end result itself. Furthermore a correlation between the two variables is often anticipated, but not actually determined to be true. In addition to these criticisms, increased flows during a flood event, and the use of sub-optimal habitat for short periods of time, further highlight the difficulty of interpreting a time series of weighted useable area (Jowett & Richardson, 1994). Whilst frustration may exist among biologists due to the inability of modelling to accurately predict fish population responses to flow changes that in light of these defects, it has suggested that it may be more helpful to regard IFIM simply as a part of a evaluative framework that is still being developed (Scott & Shirvell, 1987). It should therefore ideally be complimented by other more robust methodologies. Estimation of desired optimal biological flow values via computer modelling or other interpolation does not, therefore, dispense with biological expertise (Tennant, 1976), but instead simply provides an objective basis upon which the biologist may then interpret and present their own professional judgement and understanding of the situation (Mosley & Jowett, 1985).

Flow assessment methods such as the PHABSIM approach (represented here by the program RHYHABSIM) also rarely consider flow duration and flow variability in their analyses. Ecological repercussions of a partial day of low flow will be greatly different to those of a low flow of greater persistence e.g. through sediment accumulation (Jowett, 1996a). Occasional abstraction of such a significant volume may be of noteworthy consequence to the Waitakere River system given that invertebrate communities may still be impacted (Minshall & Winger, 1968) simply by these gross fluctuations in flows.

Furthermore, the natural regime of affected waterbodies clearly must be considered in establishing flow requirements since the resident biological population has adapted to this order (Tennant, 1976; Gordon *et al*., 1992) e.g. the Waitakere River's inanga (*Galaxias maculatus*) population are likely to be spawning in the sedges of the lower wetland during these flood events. With low flows and abstraction occurring in summer, and higher levels being reached with flush-flows occurring in the early winter, maintenance of a natural flow regime will hopefully be achieved during these ecologically significant events.

Effect of abstractions on the Te Henga Wetland

It has already been proposed that the building of the Waitakere Reservoir has resulted in a greatly reduced rate of flow through the swamp allowing vegetation growth to increase and choke even the main channel (Waitakere Ranges Protection Society, 1978). Whilst this resultant lake was still navigable as late as the 1920's when launches towed logs from both the Mokoroa Arm and the Waiti Junction to the lakes head (Waitakere Ranges Protection Society, 1978), it is presently heavily chocked with vegetation (especially introduced crack willows, *Salix fragilis*) with much of what little open water is left being difficult to negotiate even by canoe (pers. obs.). Therefore, although eroded material from the Mokoroa and Waiti Streams had already contributed significantly to the filling of the lake, a combination of catchment deforestation, exotic aquatic and amphibious plant pest introduction, illegal construction of two causeways, and the aforementioned interference with the primary water source of this lentic body have probably all been significant causes of the rapidly accelerated infilling and severe habitat degradation that is seen today (McCullough, 1998a).

Concern about the impacts of further flow reductions in the river have been voiced both by the Auckland Regional Council (Marian Jenner, Auckland Regional Council, pers. comm.), and also from reports by consulting biologists (McCullough, 1995; 1996a; 1998a) on this notable wetland. The proposed further abstraction of only another 0.004 m^3/s is unlikely to have a significant negative effect even in an especially dry season due to the large amount of catchment area this river encounters before entering the wetland. Consequently, the much larger flows when the Waitakere River reaches the wetland of which the proposed abstraction is likely to be a very small fraction. Given that about an annual mean 0.185 m^3/ s of upper catchment otherwise destined for the wetland downstream is now being diverted for other uses water (Ray Turner, Water Care, pers. comm.), any effects of this further small volume abstraction are likely to be insignificant in light of the strong negative effects already being placed on this river and wetland by this dam and its overwhelming greater magnitude of abstraction (more than 45 times as large).

Tennant (Montana) method comparison

A potential limitation to this method to New Zealand systems is the fact that its application is only relevant to streams morphologically similar to those from which the method was developed (Gordon *et al.*, 1992). This fact is of great significance when it is observed that unlike their northern hemisphere counterparts, most New Zealand rivers have poorly defined banks of a more parabolic shape (Jowett, 1996b). This is presumably due to the frequent and intense flooding that is understood to be characteristic of New Zealand's high and even rainfall climatology, and typically steep geography (Winterbourn *et al*,. 1981). The cause of this shape in the Waitakere River is likely to be at least due in part to the strong hydraulic influence of the upstream dam, and has also been noted with the Wairoa River (South Auckland) which is also contains above the reaches which are very similar in shape. Further influences are likely to come from the soft wearing nature of the catchments late sedimentary geology, and relatively large width, both helping to keep flood events contained within the main channel. The significance of this differing channel morphology, is therefore that hydraulic parameters instead tend to increase smoothly with discharge; without any clear points of inflection such as Tennant had observed with his Montana streams (Jowett, 1996a).

The Tennant method would appear to be reasonably accurately estimate the amount of total WUA available in this river at flows of 30% that would be sufficient to sustain survival for most species over a long term. This flow value of 0.0690 m^3/s is above the aforementioned points of WUA loss inflection for most species, but below the current one year return period flows currently of this river (0.0450 m^3/s). A problem with the use of historical methods in New Zealand is likely to be the assumption within them that fishes present are present because they have survived historical low flows. This is not necessarily true with New Zealand native freshwater fisheries which contain a high proportion of diadromous species (McDowall, 1990; 1993).

Conclusions and Recommendations

Sustainable utilisation of any publicly owned natural resource by a private sector must carry with it both a sense of indebtedness, and a concept of stewardship. The former should ideally help nurture the latter, and indeed will be in the interests of the applicant come the need to renew the resource consent.

Due to the low degree of WUA loss for most species following abstraction given the already highly modified nature of the river, and the presence of alternative populations and habitat in tributaries of the river, it is considered that the abstraction will not produce significantly undesirable effects on either the Waitakere River, nor the Te Henga Wetland downstream. Granting of the resource consent to abstract the proposed volume at the proposed rate will, however, require consequent implementation of measures to mitigate consequent negative effects resulting from the consequent decreased flow. Recommendations for these mitigative measures then are as follows:

1. Given that the total daily proposed abstraction is 119.0 m^3 with a proposed abstraction rate of 0.0041 m^3/s, this total daily take will be achieved in around 8 hours. Abstraction should therefore only occur at night; ideally between the hours of 11:00 p.m. and 7:00 a.m. Irrigation is more efficient during these hours with a reduced loss of water to evaporation, and increased water temperatures will largely be avoided with low flows occurring in the coolest times of the day. Lowering of the river's discharge during night hours will also mitigate the reduced sloughing and consequent possible increases in periphyton growths (Biggs & Price 1987). Apart from this, abstractions occurring the night are unlikely to have other notable effects on the periphyton population as nutrient uptake occurs in the day as well as at night, often in the form of "luxury consumption" before it is required (Barry Biggs, National Institute of Water and Atmospheric Research, pers. comm.).

2. Failure to abstract during one day should by no means constitute an authority to abstract more than the daily limit a following day. Efforts should also be made to reduce this abstraction rate by partial replacement of current grass species on the greens with varieties requiring lesser water requirements such as *Paspalum* spp.

3. Attempts must be made to reproduce a more natural riparian providing for more shade than is currently present in areas immediately downstream of the abstraction site. This will primarily involve removal of weedy herbaceous species (namely the aforementioned montbrettia *Crocosmia* x *crocosmiiflora*) and replanting with fast growing suitable ecosourced native species (Collier *et al.* 1995a, 1995b).

Further abstractions from this body of water are unlikely to be acceptable given its already highly impacted state. As Tennant (1976) so passionately argued "If water development projects cannot make it on 90% of the water carried by a stream, use of the remaining 10% probably won't justify their projects. Philosophically, it is a crime against nature to rob a stream of that last portion of water so vital to the life forms of the aquatic environment that developed there over eons of time."

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APPENDIX (A).

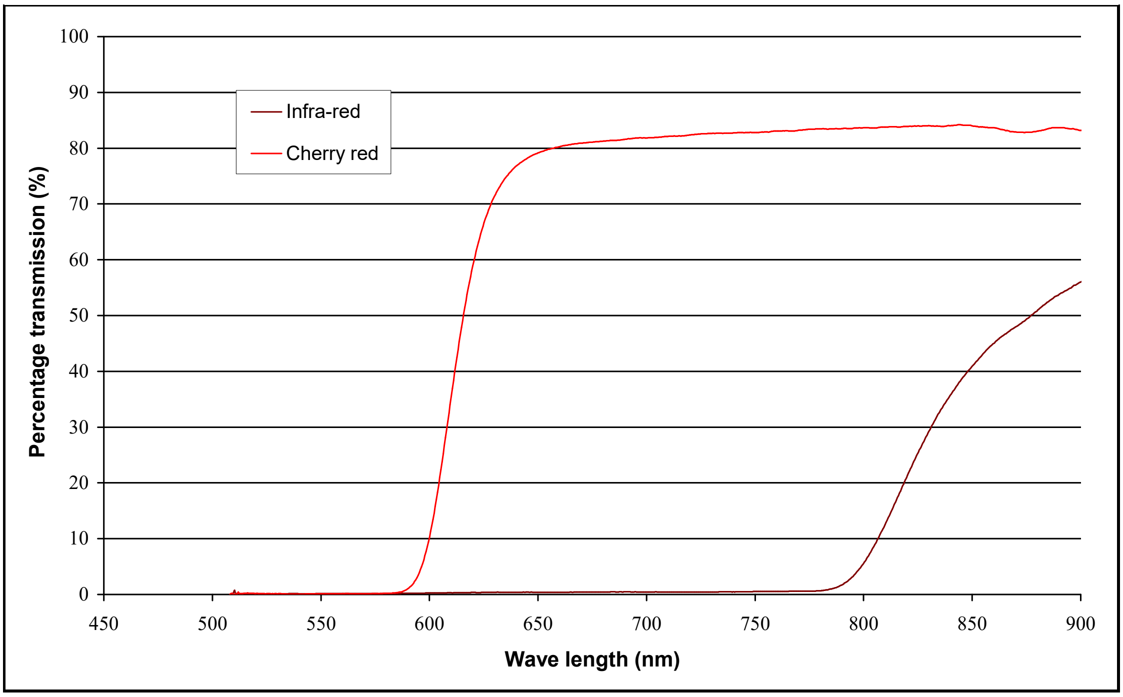

Figure 5.01/. Transmission cut-off wave lengths for the spotlight filters used in this study.

APPENDIX (B).

Table 5.02/. Field form for collection of data at night for spotlighting bias. This form can also be used for recording spotlighting counts for population estimation, with size classes further allowing for an estimation of the population's demography.

Stream name:		Water temp:		°C	
		Air temp:		°C	
Date:					
		Map no. (NZMS 260):			
		Map reference:			
Night visual counts			Reach number:		
Time start:		(hrs)	Time end:		(hrs)

Size class	Banded kokopu			Other sp.	
≤60					
61-80					
81-100					
101-120					
121-140					
141-160					
161-180					
Other sizes					
Total no.					

Table 5.03/. Field form for collecting data from the spotlight counted reaches the following day. This form can be easily modified and used with the first form to allow determination of spotlight (or other visual methodology) bias for other viable species.

Day removal electric-fishing							Date:	
Individual total lengths (mm)								
Pass 1	Time start:		hrs		Time end:			hrs
Banded kokopu					S-finned eels	L-finned eels	Other spp.	Other spp.
Total no.:								
Pass 2	Time start:		hrs		Time end:			hrs
Banded kokopu					S-finned eels	L-finned eels	Other spp.	Other spp.
Total no.:								
Pass 3	Time start:		hrs		Time end:			hrs
Banded kokopu					S-finned eels	L-finned eels	Other spp.	Other spp.
Total no.:								
Grand total:								

Physical data

5 stream widths (m)		Habitat type lengths (m); starting from downstream				
(down)1						
2						
3						
4						
(up) 5						

APPENDIX (C).

EXAMPLE OF SPOTLIGHTING ESTIMATES IN PRACTICE.

Introduction

For the purposes of an example of the spotlighting method, hypothetical data was fabricated as if spotlight counts had been made of banded kokopu in a real stream at night.

Methods and Results

Surveying was by a two-person team (one counting, the other recording), working in an upstream direction and endeavouring to make as little noise and disturbance as possible. The average width of the stream was measured at 0.3 m.

1). Spotlight counts were made for ten random 20 m reaches over a total section of 200 m. Fish seen were assigned to a total length class for each reach as per the field data sheet (Appendix (B), Table 5.01/.).

Table 5.03/. Hypothetical demographical results of a spotlighting survey of fifteen 20 m reaches over a total distance of 200 m in the hypothetical stream.

Reach number (20 m reaches)	Number of fish seen
1	14
2	10
3	16
4	7
5	20
6	18
7	15
8	15
9	7
10	22
Total	**144**
$\bar{x}$	**14.40**

2). The mean number of fish/ 20 m reach was calculated;

144 / 10 = **14.40** fish

3). The estimated actual abundance in each 20 m reach ($\overline{y}_{d,r}$) is simply the bias correction estimate ($\hat{R}$) x the mean count per 20 m reach ($\overline{x}$);

$$\overline{y}_{d,r} = \hat{R}\,\overline{x}$$

1.57 × 14.40 = **22.61** fish (average) per 20 m reach

5). The total population estimate for the 200 m reach that was sampled from ($\hat{Y}$) is found by merely extrapolating the corrected mean 20 m reach estimate ($\overline{y}_{d,r}$) by the number of reaches (N);

$$\hat{Y} = N\overline{y}_{d,r}$$

20 × 22.61 = **452** (452.20) fish total in the 200 m reach

Therefore the total abundance estimate with confidence intervals for the population of banded kokopu in this 200 m reach of the hypothetical stream is;

452 banded kokopu.

Further calculation of a density value then allows us to make direct comparisons with banded kokopu abundances in populations of other streams.

A total of 50 minutes (Figure 1.07/.) would probably have be expected to be required to spotlight count this ten 20 m reaches (a total of 200 m)

APPENDIX (D).

Table 5.04/. Field form for collecting habitat usage data from the reaches sampled by spotlight at night. This form can be easily modified and used with the following form to allow determination of habitat preferences for other aquatic species.

Habitat usage				
Night				
Stream name:		Map reference (NZMS 260):		
Date:		Temperature:		°C
Time:		Conductivity:		μScm^{-1}

Tag no.	Size class/ no. of fish	Column velocity (ms^{-1})	Surface velocity (ms^{-1})	Column depth (m)	Substrate type	Cover types present
1						
2						
3						
4						
5						
6						
7						
8						
9						
10						
11						
12						
13						
14						
15						
17						
18						
19						
20						
21						
22						
23						
24						
25						
26						
27						
28						
29						

Appendix (E)

Table 5.05/. Field form for collecting habitat availability data the following day from the same reaches sampled by spotlight at night. This form can also be readily modified and used with the previous form to allow determination of habitat preferences for other aquatic species.

Habitat Availability												
Day	Stream gauging At end of reach											
Date:		Offset:										
Time:		Depth:										
Stream name:		Mean velocity:										

Tag no.	Mean velocity (ms^{-1})	Surface velocity (ms^{-1})	Column depth (m)	Substrate type	Cover types present
1					
2					
3					
4					
5					
6					
7					
8					
9					
10					
11					
12					
13					
14					
15					
17					
18					
19					
20					
21					
22					
23					
24					
25					
26					
27					
28					
29					
30					

APPENDIX (E): 3 ½ " DISC.

This PC disk contains two folders; Habitat Preferences and Waitakere River.

Habitat Preferences folder ⇒

***.ava**: Formatted text files for available habitat for surface velocity, mean column velocity, depth, and substrate size and cover classes data for 0+, 1+, and all sizes combined.

***.use**: Formatted text files for used habitat for surface velocity, mean column velocity, depth, and substrate size and cover classes data for 0+, 1+, and all sizes combined.

***.prf**: Formatted text kernel output files for the habitat variables surface velocity, mean column velocity, and depth for 0+, 1+, and all sizes. First column = habitat variable values, second column = available habitat, third column = used habitat, and fourth column = preference.

***.xls**: Two Microsoft Excel 97 spreadsheets (Microsoft Corporation 1997) of habitat preferences for the classes of substrate size and cover type for 0+, 1+, and all sizes. These spreadsheets also contain definitions used for substrate class and cover type codes.

Waitakere River folder ⇒

Bandedk.prf: A formatted text file containing the newly developed suitability functions for banded kokopu all sizes, and 0+ and 1+ sizes. The *.prf suffix allows for immediate compatibility with RHYHABSIM.

Waitak.hab: The formatted text input file for RHYHABSIM modelling containing data for the ten cross-sections from the first sampling visit to the river.

Waitakgaug.hab: The formatted text input file for RHYHABSIM modelling containing data for the second gauging.

Tennant.xl: A Microsoft Excel 97 spreadsheet of changing WUA of all fishes and invertebrates modelled as a function of flow magnitude. Data and a chart display flows from current one-year return period flows to twice (200%) those expected prior to the Waitakere Reservoir abstraction of the Waitakere River.

Endpiece

Plate 6.01/. The author holding an especially large banded kokopu (probably female) caught under red spotlight. With a total length of over 263 mm, this fish would probably be well over 10 years old. Kokopu Stream, Royal Forest & Bird Protection Society's Matuku Reserve (West Auckland).