

The effect of terrestrial habitat fragmentation on fish populations in small streams: A case study from New Zealand

HANS S. EIKAAS, JON S. HARDING, ANDREW D. KLISKEY & ANGUS R. McINTOSH



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Habitat fragmentation edge effects on riverine fish could have implications for stream fish populations and their management. We examined effects of forest fragmentation on banded kokopu and longfin eels in New Zealand. Banded kokopu prefer small shaded streams with bouldery substrates and instream cover, whereas longfin eels are found in a variety of habitats. We hypothesized that the densities of banded kokopu would be higher in forested than grassland streams, and the densities would be lower around forest margins. We sampled pools in streams flowing from native podocarp/broad-leaf forests into grassland using Gee–Minnow traps. Pools were sampled longitudinally on each stream from >50 m into the forest to >50 m downstream from the forest margin in open grassland. Banded kokopu densities were higher in interior habitat ≥ 50 m into forests than in grassland streams. Fish caught in grassland pools were smaller than in forested pools. Longfin eels were caught in all streams, but numbers and sizes were similar in habitats; thus not likely to generate the pattern observed for banded kokopu. We found that terrestrial habitat fragmentation can impact stream fish populations by decreasing their occupancy of habitats in the vicinity of the boundary between different terrestrial habitats.

Keywords: *diadromous migration, Galaxiidae, Galaxias fasciatus, habitat fragmentation, riparian forest cover*

Hans S. Eikaas, Department of Geography, University of Canterbury, Private Bag 4800, Christchurch 8020, New Zealand. E-mail: hans.eikaas@gmail.com; Jon S. Harding, Angus R. McIntosh, School of Biological Sciences, University of Canterbury, Private Bag 4800, Christchurch, New Zealand; Andrew D. Kliskey, Department of Biological Sciences, University of Alaska – Anchorage, 3211 Providence Drive, Anchorage, Alaska, USA 99508

Introduction

Habitat fragmentation, the replacement of large areas of original continuous habitat by other habitat types, leaving isolated, disjunct habitat patches of varying sizes, can have deleterious effects on native biota (Murcia 1995). Habitat fragmentation reduces the total area and changes the configuration of original habitat, thus exposing the species that remain within the habitat fragment to the conditions of a different surrounding ecosystem, and consequently, to what have been termed ‘edge effects’. The interactions of the two juxtaposed ecosystems, often separated by an abrupt transition, or edge, can affect inhabitants of both systems. The effect of fragmentation on the remnant habitat patches and their biota is often of great conservation concern (Murcia 1995, Didham et al. 1996, Fagan 2002, Fagan et al. 1999, 2002, Fahrig 2003, Henle et al. 2004). Typically, the emphasis of edge studies has been on patterns of species richness, particularly of bird and insect communities, at habitat edges and the vegetational transition near habitat edges (Didham et al. 1998a, 1998b, Trzcinski et al. 1999, Ford et al. 2001, Lee et al. 2002, Davies et al. 2003, Kurosawa & Askins 2003, Vance et al. 2003). The net effect of fragmentation is nearly always a local loss of species within fragmented patches (Turner 1996, Turner & Corlett 1996). However, edge-mediated effects may also be evident in other organisms that do not necessarily use the fragmented terrestrial landscape per se, such as forest-stream dwelling fish. The notion that some species of fish have an affinity for elements of the terrestrial landscape, such as forests, is

well known (Barton et al. 1985, Hicks 1997, Hicks & McCaughan 1997, Hauer et al. 1999, McDowall 1990, 1997, McDowall et al. 1996, Naiman & Decamps 1997, Eikaas et al. in press), but studies of terrestrial habitat fragmentation and configuration, and in particular, the associated edge effects in stream fish communities or species are rare (Fagan 2002, Fagan et al. 2002, Eikaas 2004, Eikaas et al. 2005, Eikaas et al. in press). However, the consequences of reductions in terrestrial fragment size and connectivity of these fragments could be more severe and pose greater conservation challenges for stream fish, as dispersal of between suitable habitat fragments is restricted to existing, constrained watercourses (Fagan 2002). Thus, given the general lack of studies of edge-mediated effects of terrestrial habitat fragmentation on stream inhabitants, the effects on fish populations are likely to be of general interest as terrestrial habitats are becoming ever more fragmented.

To test whether edge effects could be detected in biota-inhabiting streams we located three streams flowing from dense native forests into open field pastures, all with similar settings with respect to underlying geomorphology and migratory distances (ease of access), known to contain populations of two native New Zealand fish: the banded kokopu *Galaxias fasciatus* and longfin eel *Anguilla dieffenbachii*, the former typically associated with small forested streams and the latter a potential predator of banded kokopu. The banded kokopu is an amphidromous galaxiid fish endemic to New Zealand (McDowall 1990). Amphidromous fishes, such as banded kokopu, migrate to sea as larvae soon after hatching and then migrate back to a

freshwater environment as post-larval juvenile fish. Adults are still common in forested streams at low altitudes and within 150 km of the sea throughout New Zealand, although banded kokopu habitat has been greatly reduced by changes in land use from native forest to pasture (McDowall 1990). The reduction in forest has been shown to severely limit the distribution of some native fish by reducing the availability of high quality habitat (Rowe et al. 1999, 2000). Banded kokopu juveniles contribute to the 'whitebait' fishery at river mouths as they re-enter a freshwater environment from the sea, although their contemporary contribution to this fishery is probably minor compared to what it once was (Rowe et al. 1999).

Adult banded kokopu are often found in pools of small order streams that have large bouldery substrates beneath a closed forest canopy (Jowett et al. 1996, McDowall 1990, Rowe et al. 2000, Rowe & Smith 2003), although in catchments where introduced salmonid fish are absent, they may appear in backwater pools of larger streams (Chadderton & Allibone 2000). Where factors such as insurmountable barriers to migratory movement and turbid water do not reduce or preclude recruitment, the presence of banded kokopu in forested streams is likely to be related to microhabitat features (Rowe et al. 2000, Richardson et al. 2001, Rowe & Smith 2003), particularly cover in pools provided by woody debris and large boulders (Rowe et al. 2000). Jowett Richardson and McDowall (Jowett et al. 1996), in a study on the relative effects of land use and instream habitat on native fish distributions, found that all banded kokopu caught in their study were associated with undercut banks or instream debris, although stream shading in itself was discounted as influencing their distribution. Thus, they concluded that banded kokopu are associated with streams that provide large amounts of instream cover. Banded kokopu also prefer low water velocities and water depths of 20–50 cm, so are common in small stream pool habitats (Main 1988, Rowe et al. 2000, Rowe & Smith 2003).

The majority of New Zealand's native freshwater fish are insectivorous, feeding mainly on stream benthos (McDowall 1990). However, some terrestrial insects are consumed by almost all native fish (McIntosh 2000) and sometimes form a major constituent of banded kokopu diet (Phillips 1926, Main & Lyon 1988, Hicks 1997). Main & Lyon (1988) also showed that over 90% by weight of the diet of banded kokopu was of terrestrial origin, comprising primarily of beetles. Thus, terrestrial land use, and particularly presence of riparian forest cover, may influence the availability of suitable prey (Jowett & Richardson 1996, Jowett et al. 1996). McDowall (1990) concluded that removal of indigenous forest cover had reduced the abundance of banded kokopu and other related species because they are rarely caught in pastoral areas. Since reciprocal subsidies between stream and terrestrial ecosystems can be important (Kawaguchi & Nakano 2001, Nakano & Murakami 2001), fragmentation of terrestrial habitats that disrupt the resource subsidies to streams could be particularly important.

Although the distribution and habitat preferences of stream fish such as banded kokopu are often well

documented (Hicks 1997, Hicks & McCaughan 1997, Rowe et al. 1999, 2000, Rowe & Smith 2003), little is known about how the configuration of the terrestrial habitat affect components of stream biota such as fish (Barton et al. 1985, Eikaas et al. 2005). We investigated whether edge-mediated effects of forest fragmentation could be observed in banded kokopu and longfin eel populations.

Methods

Study streams

We chose three streams, two of them tributaries to Kennedy Creek, and the third a tributary to Waimea Creek, located on the West Coast of South Island, New Zealand (Fig. 1). All were devoid of anthropogenic barriers to fish migrations such as weirs and dams, and all drained from dense native mixed podocarp/broad-leaf forest canopy shelter into open grassland areas. The streams supported dense populations of banded kokopu, shortjaw kokopu *Galaxias postvectis*, and the occasional koaro *G. brevipinnis* and inanga *G. maculatus*. Native redfin *Gobiomorphus huttoni* and common bullies *G. cotidianus*, were also common in these streams. Potential predatory fish in the study streams were limited to longfin eels *Anguilla dieffenbachii* and giant kokopu *Galaxias argenteus*, although introduced brown trout *Salmo trutta* were present in the main stem of the sampled stream well downstream of the study areas.

Sampling protocol

All available pools within 250 m both upstream and downstream of the forest–open interface, or edge, were sampled on three consecutive days for each stream during late January and early February 2004 using Gee–Minnow traps (approximate mesh size: 2 mm; orifice size: 5 cm) baited with Marmite, an extract of brewer's yeast (*Saccharomyces cerevisiae*), that is effective for attracting galaxiid and other native New Zealand fishes to traps (Chadderton & Allibone 2000). The traps were placed in pools just before dark (20.30 hours), and checked for fish immediately after sunrise (06.30 hours). The distance of each trap from the interface of forest-covered and open stream was measured. The numbers and fork lengths ($FL \pm 1$ mm)¹ of banded kokopu and longfin eels caught over three days were recorded to get a measure of the mean density of fish expressed as catch-per-unit-effort (CPUE) per pool and then aggregated across habitat groups. All fish were released back into the stream at the same location where they were caught following each trapping effort.

Substrate composition of each sampled pool was measured from the longest axis of 30 rocks randomly selected using the Wolman Walk method (Wolman 1954). The substrata for each pool were subsequently categorized as the percentage of mud (<1 mm in size), sand (1–2 mm), fine gravel (3–20 mm), coarse gravel (21–60 mm), cobble (61–260 mm), boulder (>260 mm), or bedrock (solid substrate), and mean substrate sizes were calculated (Table 1).

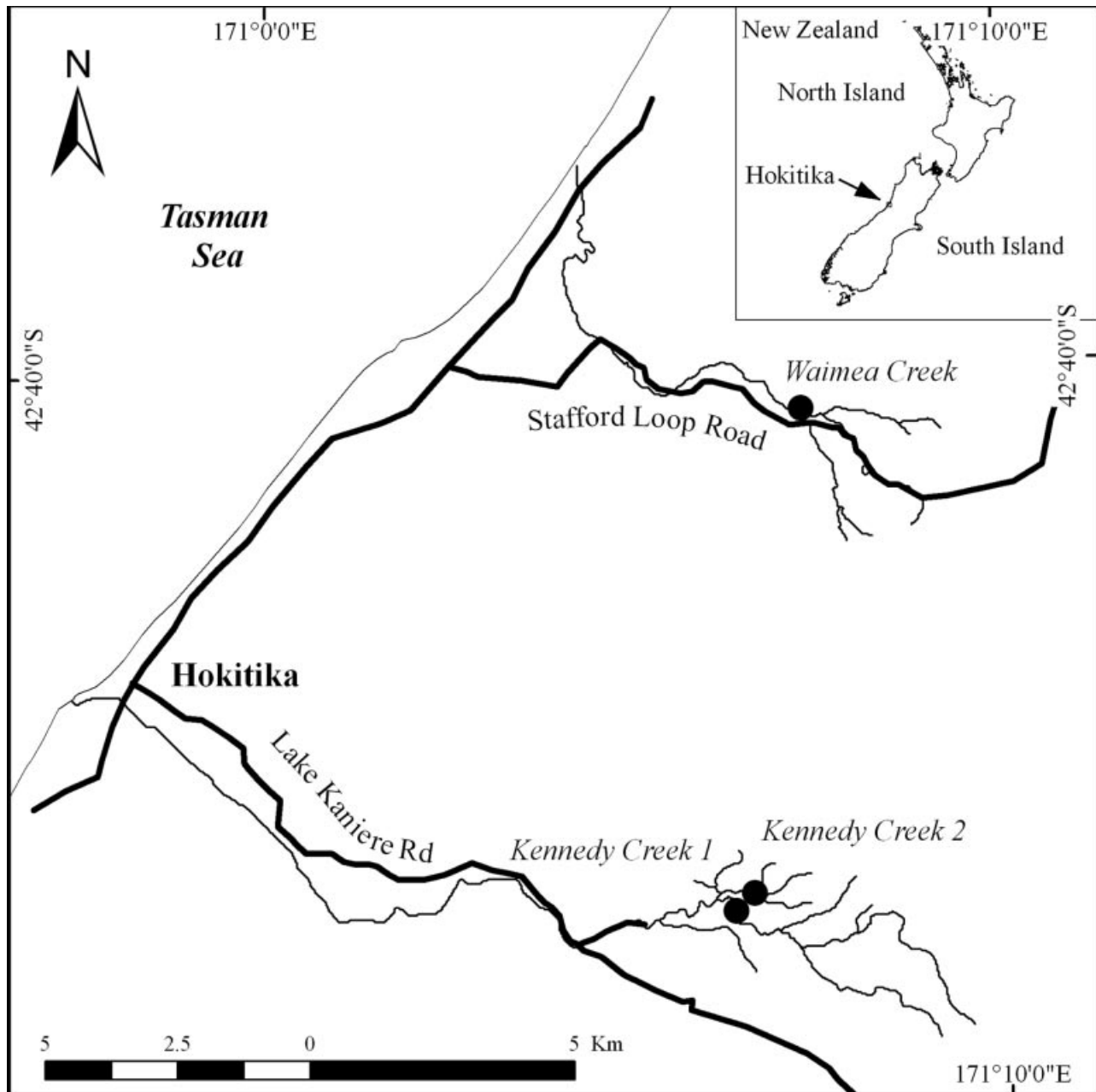


Fig. 1. Location of study streams on the West Coast of South Island of New Zealand. Solid black circles indicate location of sampled streams.

The amounts of woody material such as logs or snags present in the pools and along pools margins were recorded, and the percentage of total stream bank covered by woody material calculated. The percentage of undercut stream bank on both sides in all pools was also recorded as this pool feature may provide refuge habitat for fish.

All pools were measured in multiple dimensions to establish their relative two- and three-dimensional sizes. The widths at the upstream, middle, and downstream parts of pools were measured. Depths at 10 randomly selected locations within each pool, as well as their maximum depths, were recorded, allowing for calculations of pool volumes.

The extent of canopy cover over pools was established by taking sky-view images with a fisheye lens (Nikkor Auto 8 mm f/2.8, picture angle 180°) perpendicular to the water surface at 20 locations spaced evenly in both directions from the forest–open interface to measure any gradient in shade and canopy cover. These images were then scanned at 300 dpi into Adobe Photoshop 7.0 software and converted to black (canopy cover) and white (sky) by altering the contrast and brightness of each image. The images were then converted to binary ArcView 3.2a grids (300 dpi) with the value 1 for canopy cover, and 0 for sky, and the percentage of canopy forest cover calculated for each location.

Table 1. Physical characteristics of pools in each stream (\pm SE). Significant differences in physical characteristics between study streams were determined by ANOVA. The F and P values for these analyses are presented.

Variables	Waimea Creek		Kennedy Creek 1		Kennedy Creek 2		ANOVA	
	Mean	SE	Mean	SE	Mean	SE	F	P
Canopy shading from sky-view analysis (%)	55.06	6.53	57.44	6.83	67.83	7.074	1.02	0.366
Average substrate composition (%)								
Mud (<1 mm)	5.50	0.85	2.54	0.76	6.19	1.21	3.48	0.037
Sand (1–2 mm)	0.17	0.17	0.79	0.46	0.83	0.61	0.53	0.594
Fine gravel (3–20 mm)	14.50	0.91	12.22	1.62	14.76	1.98	0.66	0.522
Coarse gravel (21–60 mm)	19.50	2.94	12.54	1.36	14.64	1.46	3.06	0.054
Cobble (61–260 mm)	41.33	3.61	51.11	2.69	43.10	3.50	2.21	0.118
Boulder (>260 mm)	18.50	3.62	20.79	2.06	19.64	3.21	0.12	0.885
Bedrock (Solid)	0.33	0.23	0.00	0.00	0.71	0.31	2.16	0.123
Woody debris total for left and right stream bank (%)	10.84	4.53	15.87	4.63	7.18	3.59	1.13	0.330
Stream bank undercut total for left and right stream bank (%)	19.20	4.66	17.67	4.47	6.62	2.07	3.88	0.025
Pool width averages (cm)	371.00	19.25	181.11	39.57	153.21	12.59	21.96	<0.001
Pool length averages (cm)	660.83	93.65	341.35	54.69	370.71	55.87	6.23	0.003
Pool depth averages (cm)	34.00	3.42	27.68	3.92	17.01	1.83	8.74	<0.001
Pool volumes (m ³)	10.67	2.74	5.28	2.70	1.47	0.39	5.41	0.007

Data analysis

The influence of the transition from forested to open terrestrial habitat on the CPUE (catch per unit effort) of banded kokopu and longfin eels was determined by classifying sampled pools into four habitat groups: open habitat (OH, >50 m from open–forest interface), open edge habitat (OE, <50 m from open–forest interface), forest edge habitat (FE, <50 m into forest), and forest interior habitat (FI, >50 m into forest). The distance of 50 m was chosen to define ‘edge’ habitats as this is the distance frequently referred to in the literature as the extent of edge-mediated effects associated with forest boundaries (Murcia 1995). The mean differences in CPUE and fish sizes of banded kokopu and longfin eels were determined from each habitat group. Differences in CPUE and fish sizes were examined using ANOVA followed by multiple comparisons of group means by Tukey’s tests using SPSS 11.0.1.

Differences in physical habitat characteristics for pools within the four habitat groups for the three streams with respect to undercut stream banks, instream woody debris, stream shading by vegetation canopy cover, mean substrate sizes, pool depth averages, and pool volumes, were also examined using ANOVA and Tukey’s multiple comparisons tests.

Results

The CPUE of banded kokopu was significantly higher in forest habitat >50 m inside the forest margin than in the two edge habitats or open field habitat (ANOVA: $F_{3,65} = 10.75$, $P < 0.001$) (Fig. 2A). For longfin eels there were no differences in the CPUE among habitat groups, and catches were generally low for all four habitat groups (ANOVA: $F_{3,65} = 1.56$, $P = 0.208$) (Fig. 2B). The relative fork lengths of banded kokopu also differed between habitat groups (ANOVA: $F_{2,22} = 4.94$, $P = 0.004$), with open areas supporting low numbers of small fish, and the forest edge and

interior habitats both supporting larger fish of similar sizes (Fig. 3). For longfin eels there were no differences in the relative fish sizes (FL) among habitat groups (ANOVA: $F_{2,22} = 2.66$, $P = 0.100$).

Pools in open habitat had little undercutting of stream banks, whereas pools in forest core and forest edge habitat had substantial undercutting, and the forest edge habitat had intermediate amounts of undercutting (Fig. 4A). Woody debris did not differ between groups (Fig. 4B). As expected, stream shading differed significantly between forested and open areas, and although there were no significant differences in shading between the two forested habitat groups, there were significant differences in shading between open core and open edge habitats, with edge habitats being more shaded (Fig. 4C). Substrate sizes, pool depths, and pool volumes did not differ significantly between habitat groups (Figs. 4D, E, and F).

Discussion

Our results indicate that banded kokopu densities were significantly higher in native forest pools than in pools of open grassland, and increased with distance from the forest margin, thus exhibiting a response to edge habitat. An edge effect was also evident from significant differences in sizes of the fish caught among habitat groups. Larger fish were caught in forested compared to open stream pools. Fish caught within the forest edge and forest core habitats were of similar sizes, but there was greater variability in sizes of fish caught in the forest edge habitat (mean fork length \pm SE, FE: 133 ± 9.9 mm, FI, 140 ± 6.2 mm). Fish caught in the forest core habitat were consistently large, whereas more small fish were caught in forest edge habitat. Also, the few banded kokopu caught in pools outside forested stream reaches were significantly smaller than those caught within forests. In open grassland streams, there was no trend of decreasing densities of banded kokopu with increasing

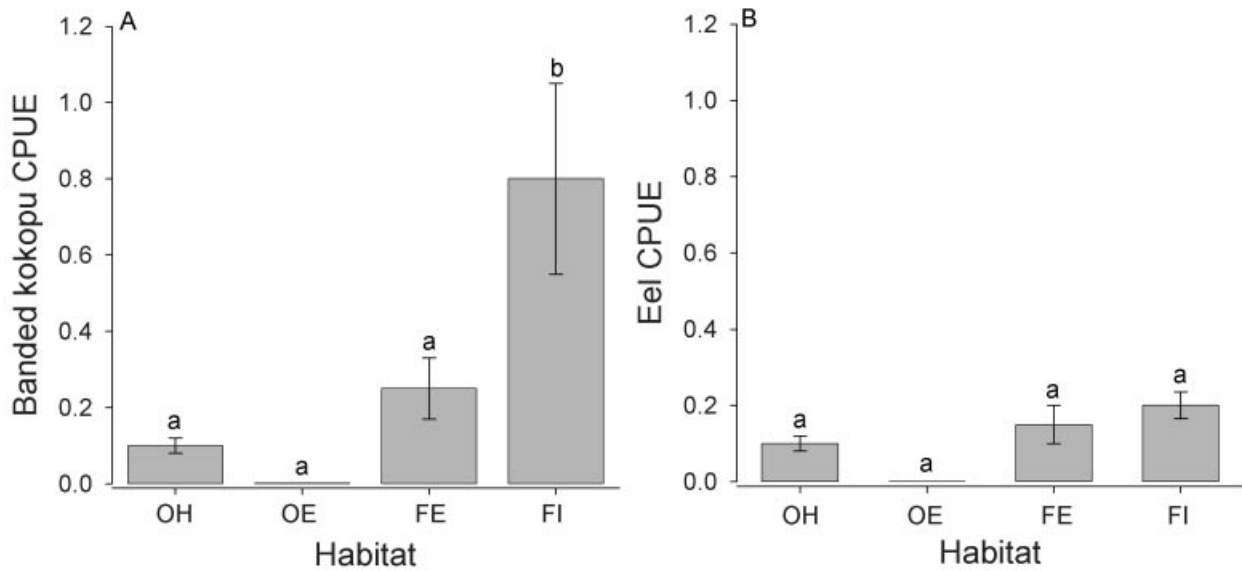


Fig. 2. Mean (\pm SE) CPUE of banded kokopu (A) and longfin eels (B) for streams in four stream habitat groupings: open habitat (OH, >50 m from forest interface), open edge habitat (OE, <50 m from forest interface), forest edge habitat (FE, <50 m into forest), and forest interior habitat (FI, >50 m into forest). ANOVA indicated a significant difference between treatments for banded kokopu ($F_{3,65} = 10.75, P < 0.001$), with different letters indicating significant differences among habitat groups (Tukey's test; $P < 0.05$), but no significant differences between treatments for eels ($F_{3,65} = 1.56, P = 0.208$).

distance into open stream habitats, giving a clear indication that large banded kokopu have a preference for forested streams.

We observed no differences in the abundance or size of longfin eels amongst the habitats in our study. However,

larger eels may not have been able to enter the Gee–Minnow traps (*c.* 5 cm diameter opening), so differences between stream habitats may have gone unnoticed. Thus, we cannot say with absolute certainty that longfin eels are not implicated in the distribution pattern observed for banded

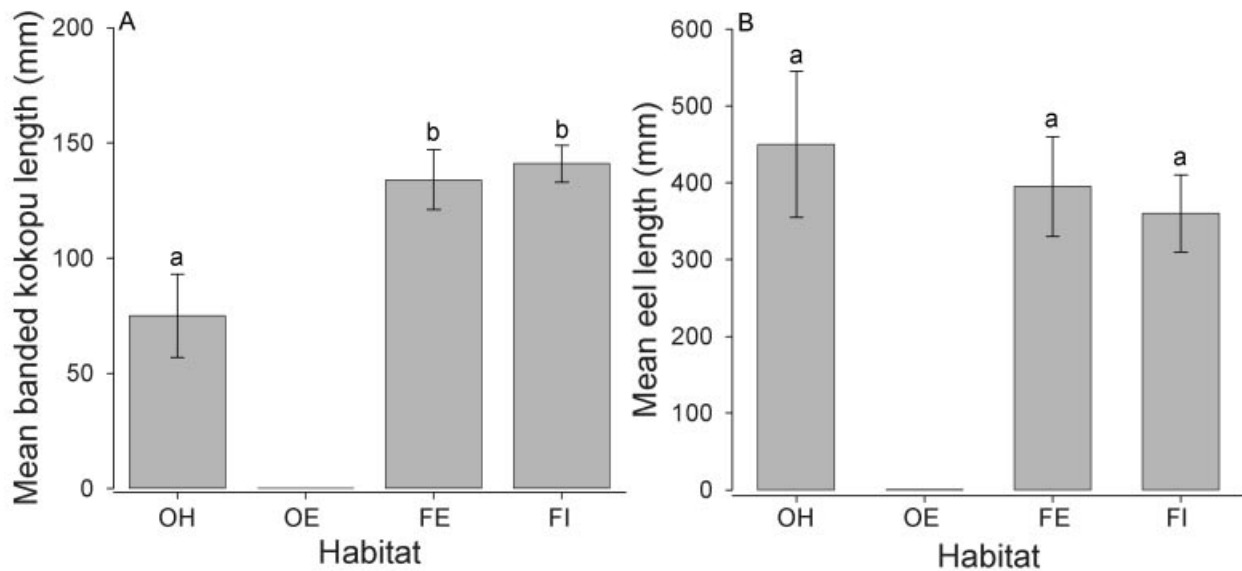


Fig. 3. Mean (\pm SD) length of banded kokopu (A) and longfin eels (B) for four habitat groupings: open habitat (OH, >50 m from forest interface), open edge habitat (OE, <50 m from forest interface), forest edge habitat (FE, <50 m into forest), and forest interior habitat (FI, >50 m into forest). ANOVA indicated a significant difference between treatments for banded kokopu ($F_{2,22} = 4.94, P = 0.004$), but no significant difference between treatments for eels ($F_{2,22} = 2.66, P = 0.100$). Different letters above error bars indicating significant differences among habitat groups (Tukey's test; $P < 0.05$).

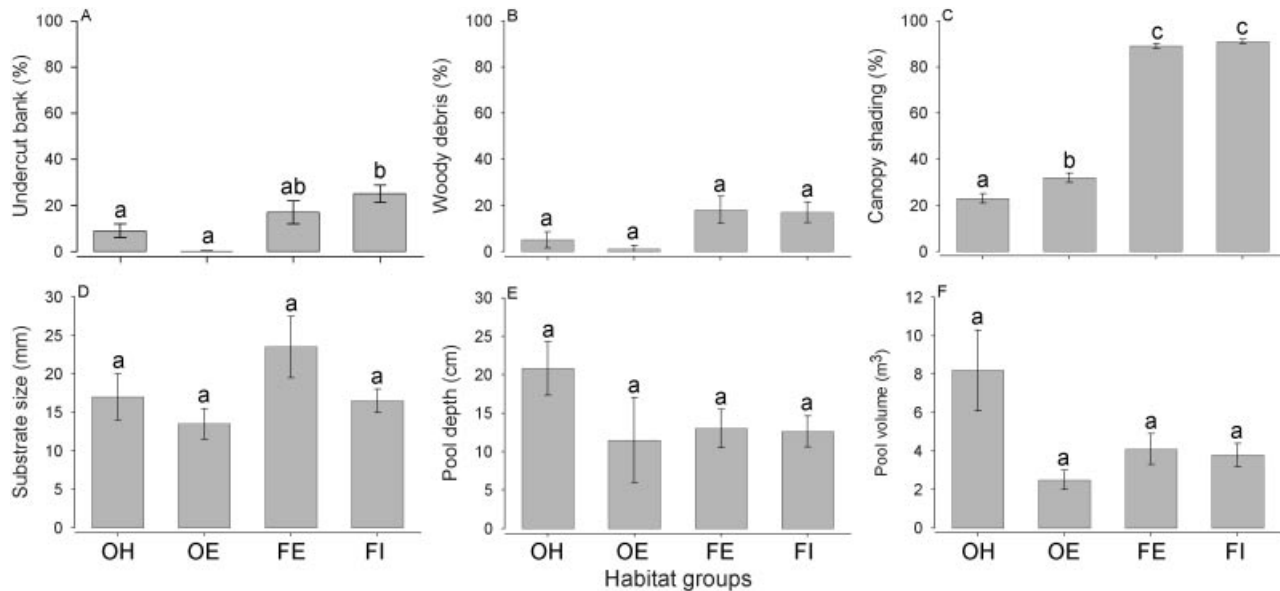


Fig. 4. Stream characteristics for the four stream habitat groupings (open habitat (OH, >50 m from forest interface), open edge habitat (OE, <50 m from forest interface), forest edge habitat (FE, <50 m into forest), and forest interior habitat (FI, >50 m into forest)) expressed as the mean values (\pm SE). Different letters above bars indicating significant differences among groups (Tukey's test; $P < 0.05$): (A) Degree of bank undercutting as a percentage of total stream banks of pools showing pools in forest core habitat have more bank cover than pools in other habitat groups (ANOVA: $F_{3,65} = 4.98$, $P = 0.004$). (B) Amount of stream banks of pools covered with woody debris as a percentage of total length of stream banks in pools, showing no significant difference among habitat groups (ANOVA: $F_{3,65} = 2.44$, $P = 0.072$). (C) Degree of canopy shading, showing pools in forests are more shaded than pools in open areas, and open edge pools are more shaded than open core pools (ANOVA: $F_{3,65} = 715.47$, $P < 0.001$). (D) Substrate sizes, showing no significant difference among habitat groups (ANOVA: $F_{3,65} = 1.39$, $P = 0.254$). (E) Pool depths were also similar for the four habitat groups (ANOVA: $F_{3,65} = 1.62$, $P = 0.193$). (F) Pool volumes were also similar among habitat groups (ANOVA: $F_{3,65} = 2.42$, $P = 0.074$).

kokopu, or that they are not affected by forest fragmentation. It may be that the larger fish of any species in the interior forested streams displace smaller fish into the inferior habitat downstream, as has been observed in populations of a closely related species, giant kokopu (David & Stoffels 2003). Besides predatory eels, which unlikely to be responsible for the patterns observed, the other fish species observed within the reaches in our study streams are small and not likely capable of displacing banded kokopu from their preferred habitats.

Studies of instream habitat use by native fish have found banded kokopu to be positively associated with bank cover and instream debris (Jowett et al. 1996, Rowe & Smith 2003). In our study, we observed the highest abundance of fish in the stream with the least amount of bank cover. Since there were no significant differences in mean substrate composition or woody debris among habitat groups, two potential refuge sources, we conclude that the presence of canopy cover is likely responsible for the observed distribution patterns of banded kokopu within our study streams. This pattern could be generated by the increased availability of terrestrial invertebrates falling into the stream from riparian forest vegetation (Edwards & Huryn 1995, 1996, Kawaguchi & Nakano 2001), thus providing increased food subsidies to banded kokopu diet in the forest interior. However, if terrestrial food subsidies were solely responsible for generating the observed pattern of banded kokopu occurrence, then one would have expected to see the food subsidy effects extending from the forest into the open habitats as invertebrates drift downstream. Another

possibility is that forested riparian cover provides refuge from avian predators.

An implication of our results for a specialist 'forest fish' such as banded kokopu is that the total length of stream affected by land use change such as forest clearance is generally longer than the length of stream directly affected by the change. Although we did not test the effects on banded kokopu of having deforested stream reaches upstream, the effects are likely to be similar, and perhaps extend further into forest patches, as the impacts on stream ecosystems often manifest themselves in a downstream direction (Pringle 2003).

Another implication of edge effects on populations of banded kokopu is that the size and shape of patches are likely to play a role. Larger, wide forest patches may have more core habitat compared to long and narrow forest patches. However, to further elucidate the significance of size and shape of forest patch impact on forest stream dwelling fish, further research on edge effects is needed, in particular with respect to increases in density of banded kokopu in streams flowing from open areas into forests. Since banded kokopu, and perhaps also other species of native fish, are affected by edge effects, reestablishment of wide forest margins around pastoral streams may significantly improve the extent of their distribution in developed catchments.

Stream biota constrained to stream channels may be affected by edges in the terrestrial cover in two ways, depending on how the edge habitat interacts with the stream. If the entire stream running through forest edge habitat is

being affected, sinuous streams would be affected more than relatively straight channelled streams as there is a greater length of stream in a given area of terrestrial habitat. Alternatively, it could be that straight-channelled and sinuous streams are affected in similar ways, if it is the along-stream distance to the edge to the juxtaposed ecosystem that is important, rather than linear distance from the stream channel.

Note

¹ The straight line distance measured from the tip of the nose to the fork of the tail of a fish.

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References

- Barton, D.R., Taylor, W.D. & Biette, R.M. 1985. Dimension of riparian buffer strips required to maintain trout habitat in Southern Ontario streams. *North American Journal of Fisheries and Management* 5, 364–378.
- Chadderton, W.L. & Allibone, R.M. 2000. Habitat use and longitudinal distribution patterns of native fish from a near pristine Stewart Island, New Zealand, stream. *New Zealand Journal of Marine and Freshwater Research* 34, 487–499.
- David, B.O. & Stoffels, R.J. 2003. Spatial organisation and behavioural interaction of giant kokopu (*Galaxias argenteus*) in two stream pools differing in fish density. *New Zealand Journal of Marine and Freshwater Research* 37, 315–322.
- Davies, R.G., Hernandez, L.M., Eggleton, P., Didham, R.K., Fagan, L.L. & Winchester, N.N. 2003. Environmental and spatial influences upon species composition of a termite assemblage across neotropical forest islands. *Journal of Tropical Ecology* 19, 509–524.
- Didham, R.K., Ghazoul, J., Stork, N.E. & Davis, A.J. 1996. Insects in fragmented forests: A functional approach. *Trends in Ecology & Evolution* 11, 255–260.
- Didham, R.K., Hammond, P.M., Lawton, J.H., Eggleton, P. & Stork, N.E. 1998a. Beetle species responses to tropical forest fragmentation. *Ecological Monographs* 68, 295–323.
- Didham, R.K., Hammond, P.M., Lawton, J.H., Eggleton, P. & Stork, N.E. 1998b. Trophic structure stability and extinction dynamics of beetles (Coleoptera) in tropical forest fragments. *Philosophical Transactions of the Royal Society of London Series B-Biological Sciences* 353, 437–451.
- Edwards, E.D. & Huryn, A.D. 1995. Annual contribution of terrestrial invertebrates to a New Zealand trout stream. *New Zealand Journal of Marine and Freshwater Research* 29, 467–477.
- Edwards, E.D. & Huryn, A.D. 1996. Effect of riparian land use on contributions of terrestrial invertebrates to streams. *Hydrobiologia* 337, 151–159.
- Eikaas, H.S. 2004. *The Effect of Habitat Fragmentation on New Zealand Native Fish: A GIS Approach*. PhD thesis, University of Canterbury, Christchurch.
- Eikaas, H.S., McIntosh, A.R. & Kliskey, A.D. 2005. Catchment- and site-scale influences of forest cover and longitudinal forest position on the distribution of a diadromous fish. *Freshwater Biology* 50, 527–538.
- Eikaas, H.S., Kliskey, A.D. & McIntosh, A.R. In press. Spatial modelling and habitat quantification for two diadromous fish in New Zealand streams: A GIS based approach with application for conservation management. *Environmental Management*.
- Fagan, W.F. 2002. Connectivity, fragmentation, and extinction risk in dendritic metapopulations. *Ecology* 83, 3243–3249.
- Fagan, W.F., Meir, E. & Moore, J.L. 1999. Variation thresholds for extinction and their implications for conservation strategies. *American Naturalist* 154, 510–520.
- Fagan, W.F., Unmack, P.J., Burgess, C. & Minckley, W.L. 2002. Rarity, fragmentation, and extinction risk in desert fishes. *Ecology* 83, 3250–3256.
- Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics* 34, 487–515.
- Ford, H.A., Barrett, G.W., Saunders, D.A. & Recher, H.F. 2001. Why have birds in the woodlands of Southern Australia declined? *Biological Conservation* 97, 71–88.
- Hauer, F.R., Poole, G.C., Gangemi, J.T. & Baxter, C.V. 1999. Large woody debris in bull trout (*Salvelinus confluentus*) spawning streams of logged and wilderness watersheds in northwest Montana. *Canadian Journal of Fisheries and Aquatic Sciences* 56, 915–924.
- Henle, K., Lindenmayer, D.B., Margules, C.R., Saunders, D.A. & Wissel, C. 2004. Species survival in fragmented landscapes: Where are we now? *Biodiversity and Conservation* 13, 1–8.
- Hicks, B.J. 1997. Food webs in forest and pasture streams in the Waikato region, New Zealand: A study based on analyses of stable isotopes of carbon and nitrogen, and fish gut contents. *New Zealand Journal of Marine and Freshwater Research* 31, 651–664.
- Hicks, B.J. & McCaughan, H.M.C. 1997. Land use, associated eel production, and abundance of fish and crayfish in streams in Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 31, 635–650.
- Jowett, I.G. & Richardson, J. 1996. Distribution and abundance of freshwater fish in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* 30, 239–255.
- Jowett, I.G., Richardson, J. & McDowall, R.M. 1996. Relative effects of in-stream habitat and land use on fish distribution and abundance in tributaries of the Grey River, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 30, 463–475.
- Kawaguchi, Y. & Nakano, S. 2001. Contribution of terrestrial invertebrates to the annual resource budget for salmonids in forest and grassland reaches of a headwater stream. *Freshwater Biology* 46, 303–316.
- Kurosawa, R. & Askins, R.A. 2003. Effects of habitat fragmentation on birds in deciduous forests in Japan. *Conservation Biology* 17, 695–707.
- Lee, M., Fahrig, L., Freemark, K. & Currie, D.J. 2002. Importance of patch scale vs. landscape scale on selected forest birds. *Oikos* 96, 110–118.
- Main, M.R. 1988. *Factors Influencing the Distribution of Kokopu and Koaro (Pisces: Galaxiidae)*. MSc thesis, University of Canterbury, New Zealand.
- Main, M.R. & Lyon, G.L. 1988. Contributions of terrestrial prey to the diet of the banded kokopu (*Galaxias fasciatus* Gray) (Pisces: Galaxiidae) in South Westland, New Zealand. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie* 23, 1785–1789.
- McDowall, R.M. 1990. *New Zealand Freshwater Fishes: A Natural History and Guide*. Heinemann Reed and MAF Publishing Group, Auckland.
- McDowall, R.M. 1997. Indigenous vegetation type and the distribution of shortjawed kokopu, *Galaxias postvectis* (Teleostei: Galaxiidae), in New Zealand. *New Zealand Journal of Zoology* 24, 243–255.
- McDowall, R.M., Main, M.R., West, D.W. & Lyon, G.L. 1996. Terrestrial and benthic foods in the diet of the shortjawed kokopu, *Galaxias postvectis* Clarke (Teleostei: Galaxiidae). *New Zealand Journal of Marine and Freshwater Research* 30, 257–269.
- McIntosh, A.R. 2000. Aquatic predator-prey interactions. Collier, K.J. & Winterbourn, M.J. (eds.) *New Zealand Stream Invertebrates: Ecology and Implications for Management*, 125–156. New Zealand Limnology Society, Christchurch.
- Murcia, C. 1995. Edge effects in fragmented forests: Implications for conservation. *Trends in Ecology & Evolution* 10, 58–62.
- Naiman, R.J. & Decamps, H. 1997. The ecology of interfaces: Riparian zones. *Annual Review of Ecology and Systematics* 28, 621–658.
- Nakano, S. & Murakami, M. 2001. Reciprocal subsidies: Dynamic interdependence between terrestrial and aquatic food webs. *Proceedings of the National Academy of Sciences of the United States of America* 98, 166–170.
- Phillips, W.J. 1926. Additional notes on New Zealand freshwater fishes. *New Zealand Journal of Science and Technology* 8, 289–298.
- Pringle, C. 2003. What is hydrologic connectivity and why is it ecologically important? *Hydrological Processes* 17, 2685–2689.
- Richardson, J., Rowe, D.K. & Smith, J.P. 2001. Effects of turbidity on the migration of juvenile banded kokopu (*Galaxias fasciatus*) in a natural

- stream. *New Zealand Journal of Marine and Freshwater Research* 35, 191–196.
- Rowe, D.K. & Smith, J. 2003. Use of in-stream cover types by adult banded kokopu (*Galaxias fasciatus*) in first-order North Island, New Zealand, streams. *New Zealand Journal of Marine and Freshwater Research* 37, 541–552.
- Rowe, D.K., Chisnall, B.L., Dean, T.L. & Richardson, J. 1999. Effects of land use on native fish communities in east coast streams of the North Island of New Zealand. *New Zealand Journal of Marine and Freshwater Research* 33, 141–151.
- Rowe, D., Hicks, M. & Richardson, J. 2000. Reduced abundance of banded kokopu (*Galaxias fasciatus*) and other native fish in turbid rivers of the North Island of New Zealand. *New Zealand Journal of Marine and Freshwater Research* 34, 545–556.
- Trzcinski, M.K., Fahrig, L. & Merriam, G. 1999. Independent effects of forest cover and fragmentation on the distribution of forest breeding birds. *Ecological Applications* 9, 586–593.
- Turner, I.M. 1996. Species loss in fragments of tropical rain forest: A review of the evidence. *Journal of Applied Ecology* 33, 200–209.
- Turner, I.M. & Corlett, R.T. 1996. The conservation value of small, isolated fragments of lowland tropical rain forest. *Trends in Ecology & Evolution* 11, 330–333.
- Vance, M.D., Fahrig, L. & Flather, C.H. 2003. Effect of reproductive rate on minimum habitat requirements of forest-breeding birds. *Ecology* 84, 2643–2653.
- Wolman, M.G. 1954. A method of sampling coarse river-bed material. *American Geophysical Union* 36, 951–956.