

Primary Research Paper

## Can forest fragments reset physical and water quality conditions in agricultural catchments and act as refugia for forest stream invertebrates?

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### Abstract

Forest fragments embedded within agricultural landscapes have the potential to provide a “forest reset effect” by mitigating agricultural effects on water quality, and acting as refugia and conservation reserves for aquatic species. We investigated the ability of forest fragments to reset agricultural effects using four catchments in the South Island, New Zealand. Two catchments were dominated by agricultural activities, but each had an isolated forest fragment in the lower valley, and two catchments had continuous riparian forest along the valley floor. Riffles sampled in continuous forest were generally deeper than those in agricultural and forest fragments, and not surprisingly streams in forest fragments and continuous forest received less light than those in agricultural land. All sites had circum-neutral pH, but both conductivity and temperature were significantly lower at continuous forest sites than agricultural and forest fragment sites. Taxonomic richness, Margalef's index and numbers of Ephemeroptera, Plecoptera and Trichoptera (EPT) taxa were significantly higher in continuous forest than at forest fragment sites, but overall invertebrate densities did not differ between fragments and continuous forest. Several taxa were abundant at agricultural and forest fragment sites, but absent or at low densities in continuous forest. They included the blackfly *Austrosimulium* spp. and two caddisflies *Pycnocentroides* sp. and *Hydrobiosis parumbripennis*. Conversely, the mayflies *Austroclima* sp. and *Coloburiscus humeralis* and the blepharicerid *Neocurupira chiltoni* were either restricted to continuous forest, or abundant in continuous forest but rare in agricultural and forest fragments. An ordination of communities separated those in agricultural and continuous forest sites, but communities at forest fragment sites were clustered among the agricultural sites. In this study forest fragments of 5–7 ha, located in the lower reaches of the catchment did not mitigate the negative upstream effects of agriculture on stream functioning. Fragment size (or riparian forest length), riparian forest width and vegetation type, and fragment location in the catchment may have critical roles in enabling forest fragments to reset the negative impacts of agriculture. Determining these characteristics of fragments has important consequences for stream remediation.

### Introduction

Landscape fragmentation in general, and forest fragmentation in particular, may be among the most important causes of habitat change, globally (Fahrig, 1997). The role of forest fragments in maintaining species diversity and as refugia for

habitat specialists has been the focus of much research in terrestrial ecosystems (Diamond, 1984; Didham, 1997; Harrington et al., 2001), and the characteristics of forest fragments have been shown to be critical for the maintenance of diversity and stable populations (Andrén, 1994; Didham, 1997; Flaspohler et al., 2001). The Theory of

Island Biogeography (MacArthur & Wilson, 1967) provided a foundation for much of this early work and focussed attention on fragment area and isolation as driving factors. Subsequent research in terrestrial environments has identified a range of characteristics that alter abiotic conditions and shape biotic responses, including area and edge effects, fragment shape, spatial and temporal isolation and habitat connectivity (Didham, 1997). The significance of forest fragments for maintaining diversity and ecosystem function in streams is a relatively unexplored issue (Storey & Cowley, 1997; Scarsbrook & Halliday, 1999). Attempts to build on knowledge gained in terrestrial systems are confounded by the continuum nature (i.e. the possible influence of upstream and downstream processes and activities) and inherent longitudinal and lateral connectivity of lotic systems.

The significance of catchment vegetation and the role of forests in controlling stream ecosystem function is a fundamental theme in stream ecology (Fisher & Likens, 1973; Vannote et al., 1980; Wallace et al., 1997). Forested streams are characterised by high allochthonous inputs, buffered water chemistries, reduced flow variation and high morphological stability, and many forested streams retain greater benthic diversity and markedly different communities than those in open, anthropogenically modified streams (Harding & Winterbourn, 1995; Wohl & Carline, 1996; Quinn et al., 1997; Townsend et al., 1997). Consequently, considerable emphasis has been placed on developing methodologies for enhancing riparian vegetation in order to restore or rehabilitate streams. Much effort has been invested in attempting to determine the ideal width and composition of riparian buffer strips, including assessments of the effectiveness of buffer strips compared to fragments of vegetation (Osborne & Kovacic, 1993; Quinn et al., 1993; Collier et al., 1995; Kiffney et al., 2004). We are interested in determining the critical physical characteristics of a forest that will enable a stream to recover from human land use impacts upstream. One simple question, for example, is how large does a fragment need to be to alter stream conditions sufficiently to support a benthic community typical of a forest? The answer to this question has direct consequences for the conservation of threatened freshwater species.

Our understanding of the effects of forest cover on streams is such that we would expect that fragments of sufficient size within an agricultural catchment should have a reset effect on a range of water chemistry and in-stream physical conditions (see Storey & Cowley, 1997; Scarsbrook & Halliday, 1999). If the fragment is large enough this “forest reset effect” should alter physical and water quality conditions. These might include altering temperature regimes by reducing summer highs and moderating winter lows, reducing sedimentation and turbidity, increasing allochthonous inputs, and shifting benthic communities to more diverse assemblages (Fig. 1). Below the forest fragment the reset effect might be expected to persist for some distance. This length of this downstream shadow might depend on several factors including; the magnitude of the reset caused by the fragment and the stream size.

The purpose of our study was to test this possible reset effect within small fragments (as compared to the downstream shadow) i.e., to determine

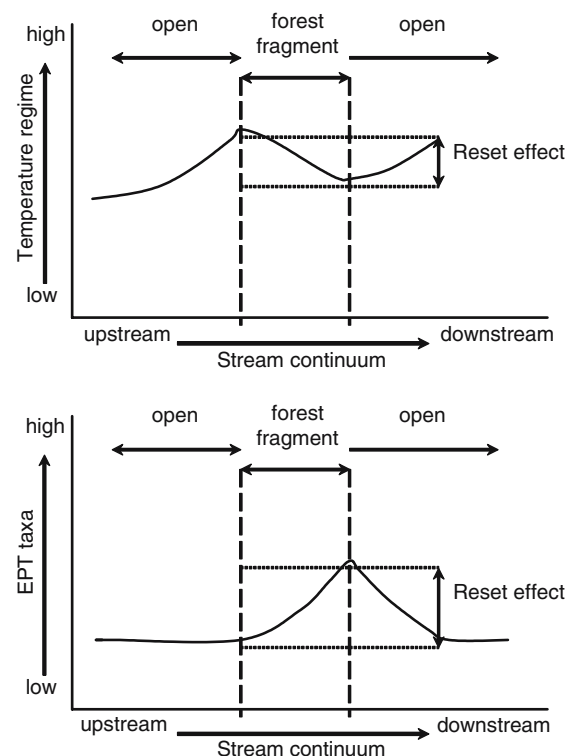


Figure 1. Hypothetical model of a potential “forest reset effect” within an agricultural stream system. EPT = Ephemeroptera, Plecoptera and Trichoptera.

if small forest fragments can mitigate upstream agricultural effects on aspects of water quality, and to compare their ability to maintain benthic stream diversity against continuously forested reaches.

## Methods

### Study area

Four catchments were sampled on Banks Peninsula, a doublet of volcanic calderas on the east

coast of the South Island, New Zealand (173° 00 E 43° 45' S) (Fig. 2). Prior to the 1840s much of the peninsula was covered in indigenous podocarp forest (Norton & Fuller, 1994), but European settlement and large-scale logging from the 1850s to 1880s resulted in removal of much of the forest. In lowland areas (< 300 m a.s.l.), the forest was dominated by totara (*Podocarpus totara* G. Benn. Ex D. Don in Lamb), matai (*Prumnopitys taxifolia* (Banks & Sol. Ex D. Don) de Laub.) and kahikatea (*Dacrycarpus dacrydioides*

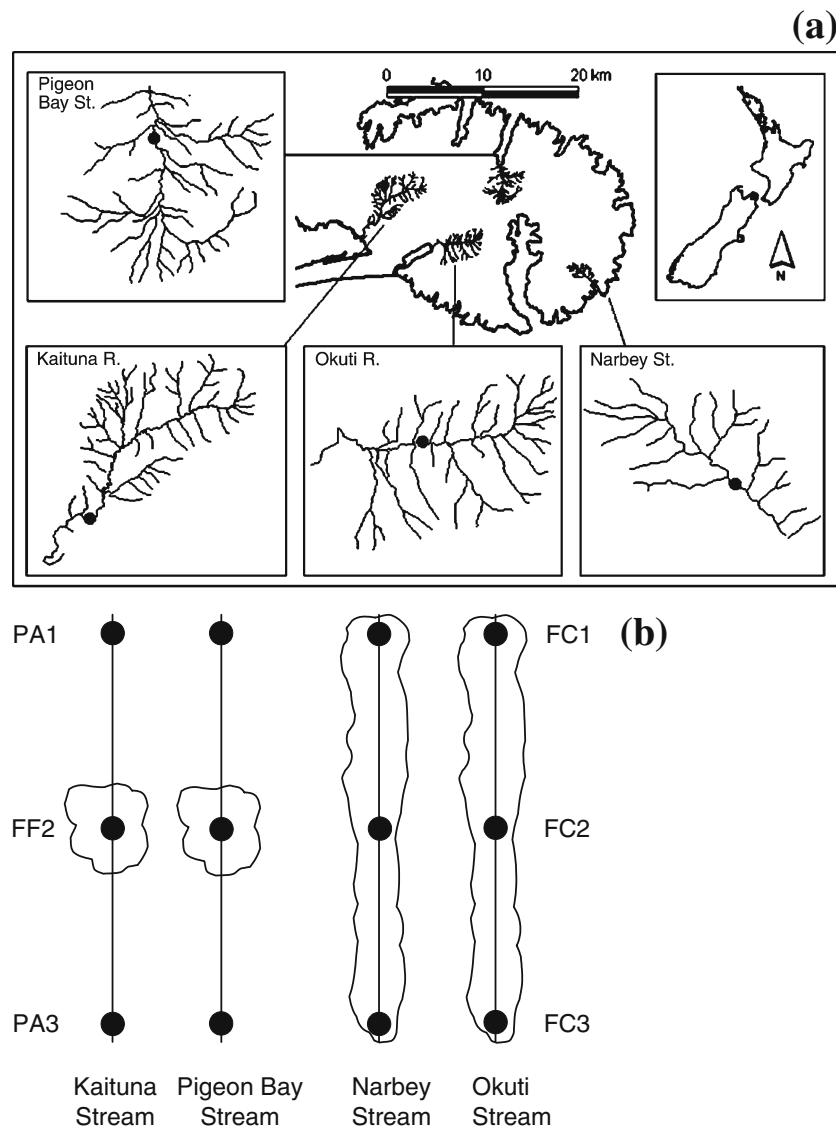


Figure 2. (a) Locations of the study catchments and study design on Banks Peninsula, New Zealand, (b) study design showing riparian vegetation, circles indicate the central study reach on each stream.

(A. Rich) de Laub.), whereas at higher elevations montane totara (*Podocarpus cunninghamii* Colenso), and New Zealand cedar (*Libocedrus bidwilli* Hook) were abundant. In 2003, over 63 regenerating and remnant indigenous forest fragments (<1 ha) were scattered throughout the region.

Rivers on Banks Peninsula are generally less than third order, short (<10 km) and steep with lowland reaches of only a few kilometres. A total of twelve sites were sampled in four catchments. Two forested reference catchments (Narbey and Okuti Streams) were dominated by continuously forested riparian zones, and two catchments (Kaituna and Pigeon Bay) were dominated by agricultural land use on the valley floors (Fig. 2). Narbey Stream was the only catchment on the Peninsula covered with almost continuous old growth forest (Table 1). In contrast, Okuti Stream had a mixture of land uses, however much of the tributaries and the upper reaches of the mainstem were in regenerating indigenous forest (Table 1). Kaituna Stream had significant forest fragments in upper tributaries (>100 m a.s.l.), but the lower 9 km of the valley supported intensive livestock farming and riparian vegetation was sparse. The only exception was Kaituna Scenic Reserve, an isolated 5 ha remnant of indigenous forest, surrounded by dairy farms. Kaituna Stream flowed through the Reserve for about 250 m. The Pigeon Bay catchment also had scattered indigenous and exotic forest in its headwaters (>100 m a.s.l.), and lower valley floor was dominated by agricultural land. Hay Reserve is an isolated 7 ha forest fragment in the lower reaches, had beef cattle farms upstream and downstream. Pigeon Bay Stream

Table 1. Characteristics of the four catchments on Banks Peninsula, New Zealand

	Forest fragment		Continuous forest	
	Kaituna	Pigeon Bay	Narbey	Okuti
Catchment size (ha)	4050	2600	1050	2600
Forest fragment size (ha)	5	7	735	88
Stream order	3	3	2	3
Elevation (m a.s.l.)	660	380	500	540
Stream length (km)	12	5.5	4.0	7.2
Stream in forest (m)	250	350	3000	4400

flows for about 350 m through the Reserve (Table 1).

Within each of the four catchments, three 30 m stream reaches about 350 m apart were selected as study sites. In the continuous forest catchments all reaches (FC1-3) had riparian forest but in the forest fragment catchments the upstream reach was in agricultural land (PA1), the middle reach was in the centre of the forest fragment (FF2), and the lower site was about 200 m downstream of the reserve, in agricultural land (PA3) (Fig. 2).

Physical and chemical factors were measured on four occasions during September and October 2002, and on two occasions benthic invertebrates were sampled. Channel morphology was characterised by measuring the number of riffles, runs and pools within each reach and their lengths. In-stream habitat conditions were estimated by measuring mean depth, width, current velocity along a transect across a riffle, run and pool within each reach (Marsh-McBirney Flowmate 2000). Substrate size was calculated by measuring the longest axis of 30 randomly selected particles, while the amount of large woody debris within each reach was estimated by measuring the length and width of each item >20 cm. Channel stability was estimated using the method described by Pfankuch (1975). Conductivity, pH and temperature measurements were made with a Oakton pH/CON10 meter, dissolved oxygen with a YSI 550 meter, and light levels with a Li-Cor LI-250 meter. Turbidity was measured from a grab water sampled returned to the laboratory and measured with a Hach 2100P meter. Chlorophyll *a* was estimated from five randomly selected cobbles at each site. Pigments were extracted in 90% ethanol for 24 h at 4 °C in the dark, and absorbances at 665 and 750 nm were measured by spectrophotometer. Chlorophyll *a* concentration was calculated as described by HACH (1990).

Benthic invertebrates were sampled using three Surber samples (0.11 m<sup>2</sup>, 0.5 mm mesh) taken from riffles at each site in both September and October. In addition, a single extensive kick-net (0.5 mm mesh) was used to collect from a wide range of habitats within each 30 m reach (e.g. boulders, gravel, sand, leaf packs, and woody debris) to add species that may not have been present in riffle samples. Samples were preserved in the field in 70% ethanol, and sorted and identified

Table 2. Mean and ANOVA results for physical parameters and water chemistry for the six land cover treatments ( $\pm 1$  SE,  $n = 2$ )

	PA1	FF2	PA3	FC1	FC2	FC3	$F_{df=5,6}$	$p$ value
Riffle length (m)	12.9 (3.9)	13.3 (3.8)	10.0 (2.5)	24.6 (2.6)	13.0 (3.0)	18.8 (6.3)	1.503	0.315
Run length (m)	9.8 (3.8)	5.5 (5.5)	12.3 (0.3)	5.2 (2.3)	11.5 (6.5)	9.0 (4.0)	0.628	0.687
Pool length (m)	7.5 (7.5)	9.0 (4.0)	7.3 (2.3)	1.8 (0.3)	5.5 (3.5)	5.0 (1.0)	0.460	0.794
Width (m)	3.2 (0.6)	5.4 (0.4)	5.3 (1.8)	4.9 (1.4)	4.3 (0.8)	3.3 (1.3)	0.806	0.584
Depth (m)	0.1 (0.0)	0.2 (0.1)	0.1 (0.0)	0.3 (0.0)	0.4 (0.1)	0.3 (0.0)	5.075	0.036
Velocity (m s <sup>-1</sup> )	0.8 (0.1)	0.7 (0.1)	0.6 (0.2)	0.7 (0.1)	0.6 (0.1)	0.7 (0.1)	0.359	0.859
Light ( $\mu\text{mol s}^{-1} \text{m}^{-1}$ )	366 (193)	18 (12)	658 (0.3)	298 (23)	84 (45)	117 (12)	10.332	0.007
Substrate size (m)	0.1 (0.03)	0.09 (0.03)	0.08 (0.03)	0.24 (0.07)	0.26 (0.02)	0.19 (0.01)	0.149	0.972
Channel stability	77.5 (4.5)	57.5 (6.5)	71.0 (4.0)	61.5 (8.5)	60.0 (5.0)	60.5 (0.5)	0.987	0.495
LWD (m <sup>3</sup> )	1.8 (0.5)	0.9 (0.3)	0.2 (0.2)	0.1 (0.1)	0.1 (0.1)	0.6 (0.2)	0.161	0.968
pH (range)	6.6–7.3	6.6–7.3	6.4–7.4	6.6–7.0	6.0–7.0	6.4–6.9		
Chlorophyll- $a$ ( $\mu\text{g cm}^{-1}$ )	4.5 (2.5)	9.2 (5.2)	13.1 (6.8)	2.3 (0.4)	1.4 (0.7)	2.1 (1.5)	2.236	0.177
Conductivity ( $\mu\text{S}_{25} \text{cm}^{-1}$ )	183 (13)	188 (9)	188 (1)	131 (4)	132 (4)	133 (5)	16.056	0.002
Dissolved oxygen ( $\text{mg l}^{-1}$ )	11.2 (0.0)	10.8 (0.2)	10.3 (0.3)	11.8 (0.2)	11.5 (0.0)	11.7 (0.1)	16.016	0.002
Turbidity (NTU)	5.4 (3.4)	6.7 (4.3)	6.2 (2.5)	5.8 (1.9)	5.8 (2.0)	6.4 (2.3)	0.041	0.999
Temperature ( $^{\circ}\text{C}$ )	11.5 (0.2)	11.8 (0.4)	11.4 (0.3)	9.4 (0.1)	9.0 (0.7)	9.0 (0.6)	8.910	0.010

PA = agriculture, FF = forest fragment, FC = continuous forest, and 1 = upper site, 2 = mid-site, 3 = lower site. PA and FF sites were in Kaituna and Pigeon Bay streams and FC sites in Narbey and Okuti streams.

in the laboratory under  $\times 8$  magnification. All individuals in the Surbers were counted and individuals identified to the lowest taxonomic level possible using available keys; usually to species or genus for insects and mollusca, and Class for Oligochaeta and Crustacea (Winterbourn, 1973; Winterbourn et al., 2000).

Comparisons of physical, chemical and biological parameters between agricultural, forest fragment and continuous forest reaches were made using One-way ANOVA (SYSTAT 10). Mean values were calculated for each parameter at each site so that sites were used as replicates rather than sampling occasions. Quantitative data were log-transformed and significance was accepted at  $p < 0.05$ . Where significant differences were found a Fisher's LSD *post-doc* test was performed. Invertebrate community composition from each sampling occasion was also examined by

non-metric multidimensional scaling (NMDS) on a Bray–Curtis similarity matrix using  $\ln(x + 1)$  transformed data (Primer 5; Clarke & Gorley, 2001). Dissimilarity values and contribution to the fauna of different land covers was analysed by SIMPER.

## Results

Channel morphology and hydrological conditions varied among reaches of agricultural, forest fragment and continuously forested streams (Table 2). The length of riffles, runs and pools did not differ significantly between reaches, and riffle width and velocity were also similar. Riffles sampled in continuous forest were significantly deeper than those in agricultural and forest fragments (Fig. 3, Table 2). Substrate size was generally smaller in

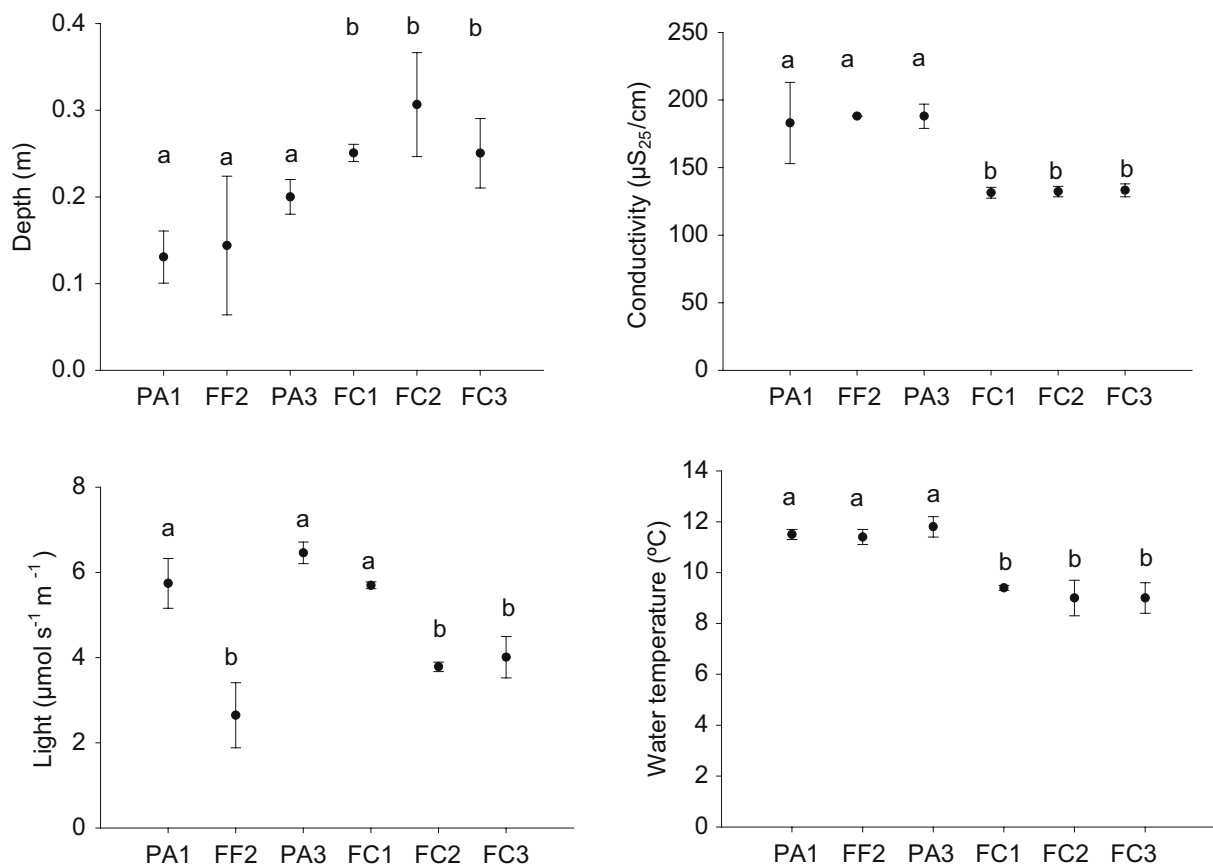


Figure 3. Significantly different water chemistry measures for agriculture, forest fragment and continuous forest reference sites ( $X \pm 1$  SE,  $n = 2$ ). PA = agriculture, FF = Forest fragment, FC = Continuous Forest, 1 = upper site, 2 = mid-site, 3 = lower site. Letters show results of Fisher's LSD tests where means with the same letter are not significantly different,  $p > 0.05$ .

agricultural and forest fragment sites than in continuously forested streams and channel stability was greater in forest fragments and continuously forested sites than in agriculture, however in both cases these trends were not statistically significant.

Water was circum-neutral in all reaches, while conductivity was consistently and significantly higher in agricultural catchments regardless of the presence of forest fragments. Of the other water chemistry parameters measured only temperature showed any marked differences which were about 2 °C lower in continuous forest compared to other reaches (Fig. 3, Table 2).

The volume of large woody debris (LWD) was greatest in upstream agricultural reaches (PA1) and least in continuous forest and in the downstream agricultural reach. However, the amounts of debris was highly variable and influenced by localised riparian willow (*Salix* sp.) trees, which

frequently shed storm damaged branches into the stream. Generally, about 50% higher light levels were recorded in agricultural streambeds than in forest fragments and some continuous forest reaches (Table 2). Periphyton biomass ranged from 100 to 600% higher in agricultural and forest fragment reaches compared to continuously forested reaches, however, no significant differences were observed across all reaches (Table 2).

A total of 63 taxa were collected from the twelve sites. Insects dominated the benthos at all reaches, with Ephemeroptera and Trichoptera accounting for 41–85% of the total numbers. Taxonomic richness was significantly higher in the continuously forested reaches (36–40 taxa), and lowest in the agricultural and forest fragment reaches (24–28 taxa)(one way ANOVA  $F_{3,12} = 4.38$ ,  $p = 0.01$ , Fig. 4). Benthic densities varied between sites, with the highest densities recorded in the upstream agricultural reaches, however, no significant

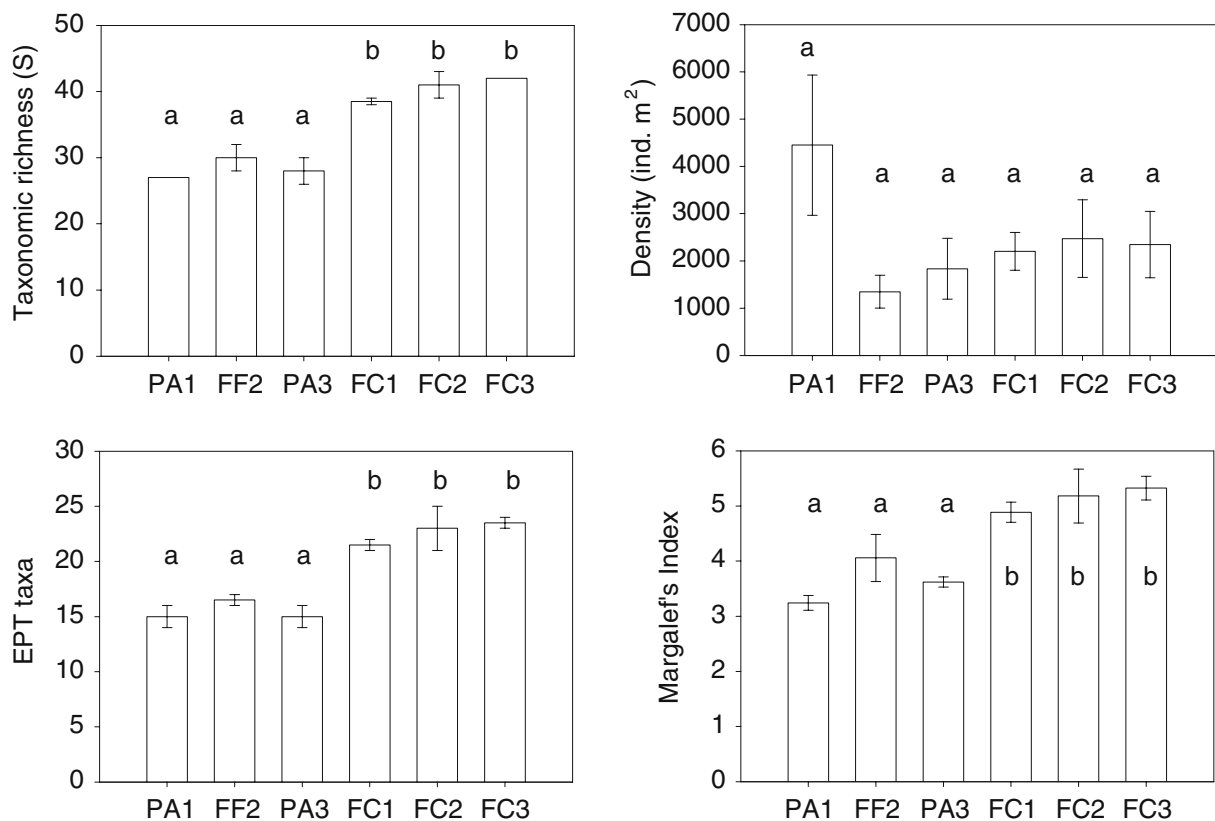


Figure 4. Diversity indices for agriculture, forest fragment and continuous forest reference sites ( $\bar{X} \pm 1$  SE,  $n = 2$ ). PA = agriculture, FF = Forest fragment, FC = Continuous Forest, 1 = upper site, 2 = mid-site, 3 = lower site. Letters show results of Fisher's LSD tests where means with the same letter are not significantly different,  $p > 0.05$ .

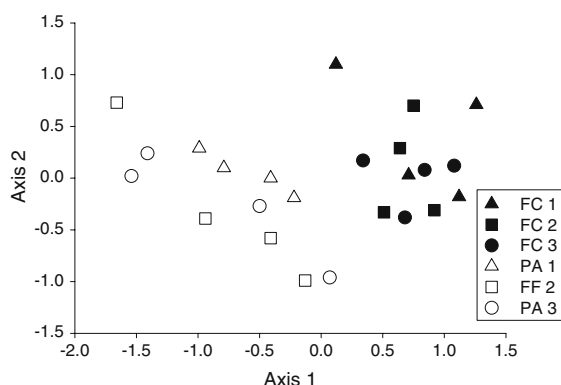


Figure 5. Non-metric multidimensional scaling for benthic invertebrate data for each site sampled in September and October 2002.

differences were detected between agricultural reaches, forest fragments and continuous forest reaches (one way ANOVA  $F_{3,12} = 0.82$ ,  $p = 0.57$ , Fig. 4). The lowest number of Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa were collected in agricultural reaches and forest fragments, and significantly more EPT taxa were recorded in continuous forest (one way ANOVA  $F_{3,12} = 6.33$ ,  $p = 0.02$ , Fig. 4). Margalef's index, which accounts for sample abundance, showed a similar pattern, with significantly higher index values at sites in continuous forest than those in agricultural reaches (one way ANOVA  $F_{3,12} = 4.72$ ,  $p = 0.04$ , Fig. 4).

Multivariate analysis of community data using NMDS clustered the reaches into two distinct groupings based on land cover. All agricultural and forest fragment reaches were clustered to the left of Axis 1, and the continuously forested reaches were to the right (Fig. 5).

A number of taxa showed marked differences in abundances depending on land cover. The blackfly *Austrosimulium*, a cased conoesucid caddis *Pycnocentroides*, and a predatory hydrobiosid caddis *Hydrobiosis parumbripennis* McFarlane were frequently abundant in agricultural and forest fragment reaches, but occurred in very low numbers in continuous forest (Fig. 6, Table 2). In contrast, the Banks Peninsula endemic blepharicerid *Neocurupira chiltoni* (Campbell), the mayflies *Austroclima jollyae* Towns & Peters, and *Coloburiscus humeralis* (Walker) were absent or in low numbers in agricultural and forest fragment reaches, but present (and in some cases relatively

abundant) in all continuous forest reaches (Fig. 6, Tables 2 and 3).

## Discussion

Historically, the widespread felling of ancestral forest undoubtedly had far reaching consequences for terrestrial and aquatic species adapted to forested ecosystems. Habitat loss is widely recognised as one of the most important threats to indigenous freshwater species (Allan & Flecker, 1993; Benstead et al., 2003). Benstead et al. (2003) suggested the high endemism of Madagascar streams is under threat from deforestation and by inference local habitat loss, and Bojsen & Jacobsen (2003) concluded that deforestation within the Ecuadorian Amazon has resulted in declines in benthic diversity as reductions in litter inputs to streams and a reciprocal rise in periphyton biomass has occurred. Within New Zealand, Harding (2003) has shown that the distribution of several aquatic species, endemic to Banks Peninsula, are now seemingly restricted to forested streams. In general, our understanding of the effects of forest fragmentation on terrestrial ecosystems is relatively well developed (Didham, 1997). Much work has been done on fragment size, shape, and edge effects and the response of multiple species to these factors. In contrast, the study of landscape fragmentation and the response of aquatic systems to it are much more poorly understood. In lotic systems the distribution and dispersal of species, and water and habitat quality are strongly influenced by both the longitudinal and dendritic network of streams, and these factors add further complications to understanding fragmentation effects.

Streams in continuous forest in the present study were morphologically similar to the agricultural and forest fragment stream reaches, differing primarily in riffle depth and in-stream light levels. Although we did not measure FPOM and CPOM in our study we might expect them to differ also between agricultural, forest fragment and continuously forested reaches. The differences in physical variables we detected were not surprising, as agricultural catchments might be expected to generate greater sedimentation, and thus show reduced depths and mean substrate size. Interestingly, the differences in in-stream light levels did

not transfer into differences in algal biomass between forested, forest fragment and agricultural sites. Comparable agricultural and forested algal biomass levels have been noted in other studies (Quinn et al., 1997; Scarsbrook & Halliday, 1999), and may be explained partially by the dappled light conditions, and stable large substrate in many New Zealand indigenous forested streams that

enable patchy concentrations of low light adapted algae to persist.

The benefit of forest fragments as a refuge for forest stream specialists is an issue of increasing importance as landscape fragmentation continues globally. Heartsill-Scalley & Aide (2003) state that forested riparian areas in nonforested landscapes can assist in the maintenance of diversity and act

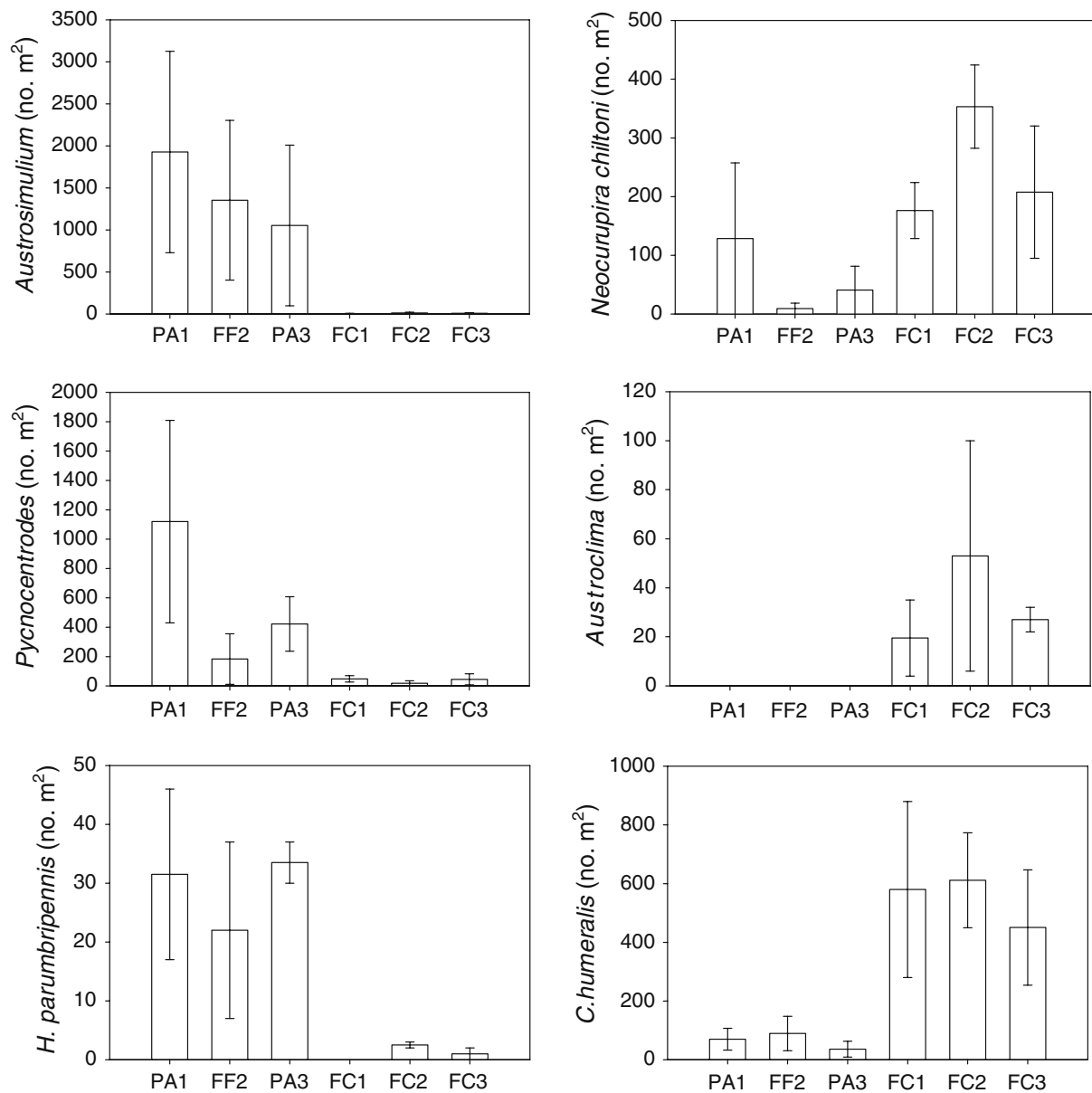


Figure 6. Abundance of selected taxa in agricultural, forest fragment and continuous forested sites ( $X \pm 1$  SE,  $n = 2$ ). PA = agriculture, FF = Forest fragment, FC = Continuous Forest, 1 = upper site, 2 = mid-site, 3 = lower site.

Table 3. Results of SIMPER analysis of mean dissimilarity values, using abundance data, identifying the five most important taxa that contribute to observed assemblage differences between forest fragment, continuous forest and agricultural reaches

Taxa	Mean abundance				
	Fragment	Forest	Dissimilarity	Contribution (%)	Cumulative (%)
<i>Austrosimulium</i> spp.	1353.7	8.0	3.5	8.3	8.3
<i>Neocurupira chiltoni</i>	9.3	245.7	2.6	6.1	14.4
Hydraenidae	0.0	65.3	2.5	5.8	20.2
<i>Zelandoperla decorata</i>	0.9	58.2	2.1	5.0	25.2
<i>Austroclima</i> spp.	0.0	33.2	1.8	4.2	29.4
	Agriculture	Fragment	Dissimilarity	Contribution (%)	Cumulative (%)
<i>Pycnocentroides</i> spp.	770.8	183.3	2.2	6.8	6.8
<i>Pycnocentria evecta</i>	515.3	163.0	1.8	5.7	12.6
<i>Neocurupira chiltoni</i>	85.2	9.3	1.7	5.3	17.9
<i>Hydrobiosis parumbripennis</i>	32.4	22.2	1.5	4.7	22.6
<i>Aoteapsyche</i> spp.	196.8	138.0	1.5	4.7	27.3
	Agriculture	Forest	Dissimilarity	Contribution (%)	Cumulative (%)
<i>Austrosimulium</i> spp.	1490.7	8.0	3.2	7.8	7.8
<i>Pycnocentroides</i> spp.	770.8	37.2	2.2	5.6	13.4
<i>Neocurupira chiltoni</i>	85.2	245.7	1.8	4.6	17.9
<i>Hydrobiosis parumbripennis</i>	32.4	1.4	1.7	4.3	22.2
<i>Coloburiscus humeralis</i>	44.0	547.3	1.7	4.2	26.4

as refugia for mature forest species. In preparing this paper we extensively searched the literature to identify stream studies which supported this assertion and demonstrated the recovery of stream communities in modified landscapes due to forested patches. We were surprised by the paucity of studies on this topic. In this study two forest fragments ranging from 5 to 7 ha in size, located in the lower reaches of two third-order agricultural river systems provided habitat of marginal quality for forest stream specialist invertebrates. These forest fragments, which provided riparian inputs along 250–350 m of stream provided little or no mitigation of water quality conditions (temperature, dissolved oxygen, conductivity or turbidity) and had no detectable effect on the physical characteristics or morphology of the streams. Taxonomic richness, though slightly higher within the forest fragments did not differ from that recorded in surrounding farmland stream reaches. Furthermore, these fragments supported few differences in community composition from the agricultural reaches. Scarsbrook & Halliday (1999) investigated three first-order streams, which passed from pasture into indigenous forest. They found that several characteristics of stream morphology

(e.g., channel width and velocity) returned 50 m into the forest to similar values recorded in a native forest reference stream. However, water quality conditions, such as, dissolved nitrogen and water clarity did not recover to forested stream levels after 300–350 m. In contrast, benthic invertebrate communities had returned to forested stream diversity and density by 300–350 m. Similar findings were reported by Storey & Cowley (1997) who assessed the changes in water chemistry and benthic communities in three pastoral streams that entered remnant forest. They found that water quality conditions improved relatively slowly as streams moved from pasture to forest, while benthic communities recovered significantly 600 m into the forest.

In this study taxonomic richness was highest at continuously forested sites, and considering mixed podocarp forest dominated these catchments for about 20,000 years (from the Pleistocene Glaciation) it is perhaps not surprising that more species have adapted to conditions in a forested stream environment than an open one. Forest clearance on Banks Peninsula during the 1840s–1880s radically transformed the landscape, and must have triggered marked changes in in-stream conditions

regionally. Internationally, the effects of afforestation, and production forestry logging on the physical, chemical and biological components of stream ecosystems have been widely documented. Numerous studies have noted that benthic communities recover from logging disturbance over many years, and some literature indicates that benthic diversity may even return to pre-logging levels. However, it seems likely that region-wide deforestation as experienced on Banks Peninsula where 99% of indigenous forest was removed would have resulted in severe restrictions in the distribution and population viability of species adapted to forested stream habitats. Currently several species endemic to Banks Peninsula have restricted distributions limited to indigenous forested stream reaches (Harding, 2003).

#### *How effective are forest fragments as refugia for forest stream species?*

The two forest fragments investigated in this study did not provide sufficient habitat to mitigate the effects of agricultural activities within their respective catchments. Both taxonomic richness, and numbers of EPT taxa were significantly lower in the forest fragments than at the continuously forested sites, and diversity in the forest fragments differed little from that recorded upstream and downstream in agricultural land. The inability of these fragments to mitigate agricultural effects (a “forest reset effect”) and support higher diversity is probably a consequence of several landscape-scale factors related to the magnitude of agricultural activities and inputs within the catchments, overall stream size (these were third-order streams), and characteristics of the forest fragment which might include, the total length of the riparian zone in tall vegetation, the condition of the riparian zone (e.g., vegetation age, and species diversity), the location of the vegetation fragment within the catchment, and the degree of isolation from other significant forest fragments. Despite the likely importance of all these factors, many of the studies on the effects of fragmentation on terrestrial species have shown that fragment size can be a critical “first measure”. How useful forest fragment size is for assessments of the responses of stream communities is unclear and represents a challenge for stream ecologists.

If freshwater ecologists are to make a contribution to the conservation and preservation of threatened freshwater species, then studies which quantify and elucidate the necessary characteristics of forest fragments (and riparian buffer zones) should be a priority for future research. Currently, water managers and conservation agencies are hindered by a lack of robust science to answer these questions.

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