

Transition from pasture to native forest land-use along stream continua: effects on stream ecosystems and implications for restoration

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INTRODUCTION

Pastoral and production forestry land uses are perceived to be the main causes of degradation of New Zealand's inland waters (Smith 1993). Indeed, the impact of land use on the health of streams has been described as the most significant environmental issue facing our nation (Upton 1994). A number of studies in New Zealand have demonstrated effects of differing land use on stream hydrology (Smith 1993), water chemistry (Maasdam & Smith 1994), light and temperature regimes (Davies-Colley 1997; Rutherford et al. 1997), energy sources (Quinn et al. 1997; Townsend et al. 1997), and aquatic biota (Harding & Winterbourn 1995; Quinn et al. 1997; Townsend et al. 1997). In general these and other studies have assessed differences between catchments of differing land use, rather than effects associated with land-use change within a catchment. Although these studies have highlighted the effects of pastoral land use on stream ecosystems, resource managers (e.g., regional councils) and resource users (e.g., farmers) also require information on the means of mitigating the adverse effects of pastoral land use.

One of the most effective ways of protecting stream ecosystems from the effects of pastoral land use is through improved management of riparian zones (Quinn et al. 1993; Collier et al. 1995). Riparian areas are zones of direct interaction between terrestrial and aquatic ecosystems (Gregory et al. 1991), and can provide buffers for stream ecosystems against bank erosion, contaminant input, high light and temperature levels, excessive in-stream primary production, and low levels of allochthonous input (Quinn et al. 1993). Prior to human habitation (c. 1150 AD) most of New Zealand's land area was covered in native forest (Collier et al. 1995). This suggests that to restore in-stream conditions towards a more "natural" state, reforestation of riparian zones is required.

Regardless of the environmental benefits, it is seldom economically feasible to retire whole catchments from pastoral production and in New Zealand it is often impracticable to plant extensive riparian

Abstract Three first-order, hill country, pasture streams in Waikato, New Zealand, were chosen to investigate the effects of patches of late-succession indigenous riparian forest on water quality, epilithon, stream morphology, and aquatic macro-invertebrates. Sites were situated in open pasture and at two distances (c. 50 and 300 m) into a forest remnant on each stream. Shade, channel width, and epilithon biomass were restored to conditions similar to a native forest control site within 300 m of the streams entering the native forest remnants, whereas water chemistry and levels of surficial fine sediment changed more slowly. Invertebrate community composition showed shifts towards the native forest condition just 50 m into the forest remnants, and full recovery had occurred within 300 m. Results from this study suggest that discontinuous restoration of riparian margins could mitigate some changes associated with pastoral land use, but sediment and water quality problems may not be solved.

Keywords stream habitat; forest remnants; channel morphology; water quality; benthic macro-invertebrates; riparian restoration; New Zealand

zones throughout a catchment because of patterns of land ownership. One option is to restore discontinuous patches of riparian vegetation, or protect remaining patches of intact forest. Such an approach can only be effective if the patches are large enough to provide stream conditions necessary for recovery to near the natural state. However, the distance required to restore native conditions will vary depending on the variable being targeted for restoration (Storey & Cowley 1997). For example, reduction in light levels reaching the streambed could be achieved over relatively short distances, whereas restoration of a forest stream temperature regime may require hundreds of metres of riparian restoration (Rutherford et al. 1997).

Our study was conducted within an on-going research programme, the aims of which are to provide a better understanding of the effects of land use on stream ecosystems and to develop practical mitigation strategies (Quinn & Cooper 1997). The programme centers on long-term ecological research (LTER) sites in and adjacent to Whatawhata Research Station, near Hamilton, New Zealand. There are several sites in the area where headwater streams run abruptly from pasture into patches of native forest. These transition zones provide "natural

experiments" on the effects of late-succession indigenous forest in riparian areas of pastoral streams. The aim of this study was to compare stream physical habitat, water quality, and aquatic biota in stream reaches above and at increasing distances below these pasture-native forest transition zones.

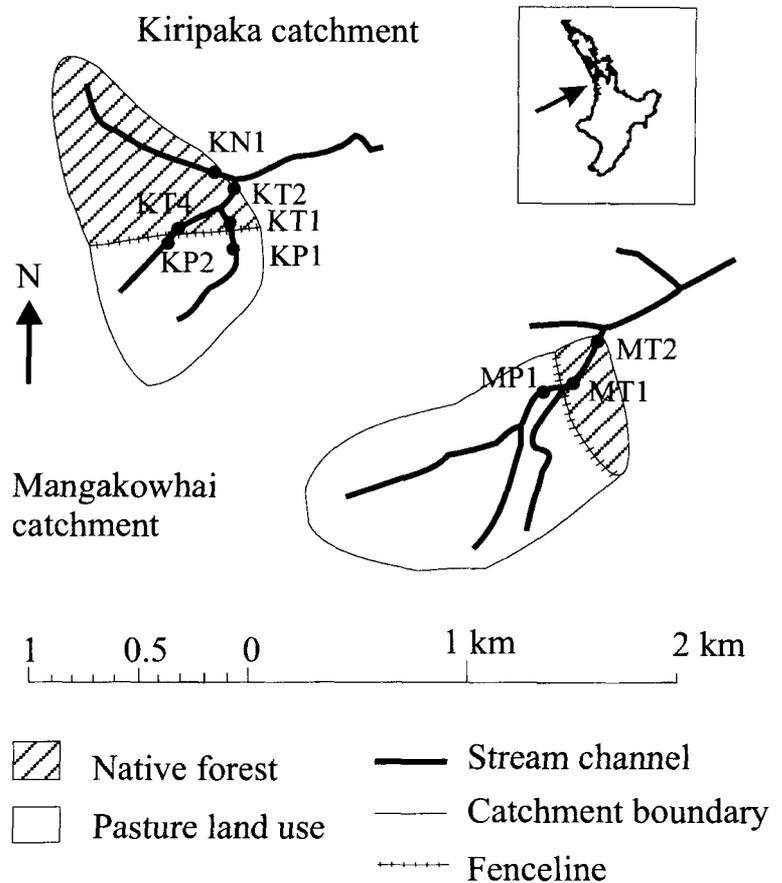
The effects of land use on stream ecosystems at Whatawhata have been described, in detail, by Quinn et al. (1997). The differences between pasture and native forest stream ecosystems observed by Quinn et al. (1997) form the basis of *a priori* predictions tested in the present study (Table 1). We predicted that: (1) local effects of pastoral land use (such as light climate and epilithon biomass) would be restored to the native forest condition within a short distance (c. 50 m) of entry of the streams into forest remnants; (2) several parameters would not be restored until a stream had flowed through several hundred metres of native forest (e.g., suspended and bed sediment levels and stored coarse particulate organic matter (CPOM); and (3) stream invertebrate assemblages would be less diverse immediately below the transition zone than in either pasture or native forest situations as a result of poor habitat quality and the low levels of both algal epilithon and allochthonous food sources.

Table 1 Changes in key stream habitat and biotic variables relative to native forest conditions. Information on pasture comparison comes from Quinn et al. (1997), whereas *a priori* predictions of change with varying distance into native forest remnants are tested in the present study. (CPOM=coarse particulate organic matter; DOC=dissolved organic carbon.)

Variables	Pasture	0–50 m into forest	300–350 m into forest
Morphology			
Shade	reduced	restored	restored
Channel width	narrower	increased	restored
Current velocity	increased	decreasing	restored
Bed sedimentation	increased	increased	restored
Woody debris	reduced	*	*
Water quality			
Temperature	increased	*	*
Dissolved nitrogen	increased	increased	restored
Clarity	reduced	reduced	restored
Carbon resources			
Stored CPOM	reduced	increasing	restored
DOC	increased	increased	restored
Algal epilithon	increased	restored	restored
Invertebrates			
Taxonomic richness	no change	reduced	restored
Sensitive species	reduced	reduced	restored
Density and biomass	increased	reduced	restored

*Variables not tested in this study.

Fig. 1 Map showing the location of the study sites on tributaries of the Waipa River, New Zealand. First letter in the site code is for the parent stream (Kiripaka or Mangakowhai), the second is for the land use around the reach (P for a pasture, T for transition forest, or N for native forest).



MATERIALS AND METHODS

Study area

Two catchments (Kiripaka and Mangakowhai) in the Hakarimata Ranges west of Hamilton, New Zealand, ($175^{\circ}5'E$; $37^{\circ}47'S$) were chosen because they contained one or more stream branches with a pasture headwater catchment that moved into a late succession forest at a fenceline (Fig. 1). The forest was composed of mostly broadleaf species and was estimated to be 60 years old from aerial photos. The oldest aerial photo (1943) showed the transition zones in dense scrub, suggesting they had been cleared of forest at an earlier time. The study sites were on first-order or second-order permanently-flowing cobble streams, with steep catchments of less than 1 km^2 .

Nine 50 m reaches were selected within the two catchments (Fig. 1). Site KP1 began in pasture 70 m above the fenceline to avoid a series of steep bedrock

sections. KT1 began in forest 30 m below the fenceline to avoid an area of open bush and a series of waterfalls. The other pasture and transition zone reaches began 10 m above or below the fenceline. The transition zone site at KT2 was 200 m below KT1, and 300 m below KT4. None of the streams had significant tributaries within the study reaches. The catchment for KN1 was entirely in native forest. A small tributary entered the Mangakowhai between MP1 and MT1, just above the fenceline. This tributary was slightly smaller than MP1, it had the same catchment land-use and did not appear to differ in habitat characteristics. MT2 was 200 m down stream of MT1. The Mangakowhai catchment did not have a native forest reference site, however KN1 had a similar hydrology, catchment area, and forest type.

Sampling at the nine sites was carried out under stable flow conditions between 28 November 1995 and 10 January 1996.

Physical and chemical sampling

The light environment of the sites at stream level was quantified by comparing diffuse non-interceptance light (DIFN) at 20 points along the stream reach with a nearby unshaded hilltop reference site using a pair of LAI-2000 canopy analysers. These instruments quantify riparian and topographical shade by measuring the amount of sky in five concentric rings viewed through a fisheye lens (Welles & Norman 1991). From this an index of light exposure can be calculated, expressed as diffuse non-interceptance (DIFN) (Davies-Colley & Payne 1998). Channel slope was measured for each of the sites over the 50 m reaches, using a surveyor's level and survey pole. Channel slope was also measured between sites KP1 and KT1. Physical measurements of stream morphology were taken at 20 evenly spaced cross-sections along the study reaches. At each cross-section water, channel and bankfull widths were measured, and the depth of left and right-hand bank undercuts were also measured. At each cross-section water depths were taken at five evenly spaced points, the percentage of overhanging riparian vegetation and coverage of macrophytes were estimated and c. 10 randomly selected sediment particles were measured (intermediate axis) across the width of the stream (particle size classes: <2 mm; 2–8 mm; 8–16 mm; 16–32 mm; 32–64 mm; 64–128 mm; 128–256 mm; >256 mm; small wood diameter <10 cm; large wood diameter \geq 10 cm). At the even numbered cross-sections velocity profiles (three measurements evenly spaced across the channel where water depth allowed) were recorded using a Montedoro-Whitney (Model PVM2A) current meter.

Stored suspendible inorganic sediments (SIS) and benthic particulate organic matter (BPOM) in the surface sediments were sampled at seven randomly selected transects in each study reach. Bed sediments were stirred with a pole in an enclosed cylinder (diameter 23 cm) to a depth of 5 cm in a known volume of stream water (determined from mean water depth measured at 10 points). When the suspended fines were fully mixed, a 120 ml sample was taken for dry mass (DM, 104°C to constant mass) and ash-free dry mass (AFDM, 400°C for minimum of 6 h) analysis. The mass of SIS and BPOM per square metre of streambed surface was estimated from the area of bed the cylinder covered and the water depth (allowing for the background DM and AFDM in the water).

A 1-litre water sample was taken at either end of the reach and the samples were analysed in the

laboratory for pH (Radiometer 26 meter), electrical conductivity (EC, Radiometer CDM83 meter), turbidity (nephelometric turbidity units, NTU; Hach 2100A meter), and suspended solids (SS; GF/C filtration and 24 h at 105°C, APHA 1989). A sub-sample of the water was filtered and analysed for: dissolved reactive phosphorus (DRP; automated molybdenum blue/ascorbic acid colorimetry, APHA 1989); ammonium nitrogen (NH₄-N, automated phenol/hypochlorite colorimetry method, APHA 1989); nitrate/nitrite nitrogen (NO₃/NO₂-N, APHA 1989); dissolved organic carbon (DOC; alkaline persulphate with UV photo-oxidation and CO₂ detected by infrared gas analyser); filtered absorbance at 440 nm; and alkalinity (mg CaCO₃/litre).

Channel stability over each of the study reaches was evaluated visually by a single observer using the Pfankuch stability index (Pfankuch 1975; Collier 1992). The Pfankuch index scores 15 variables in three regions of the stream channel (upper and lower banks, and streambed) with high scores indicating unstable conditions at the reach scale. Catchment size was calculated by drawing a polygon around the estimated catchment boundary and converting the area of the polygon into km² using the Arc/Info Geographical Information System.

Biological sampling

Seven randomly selected stones were collected from runs at each site in order to estimate periphyton biomass. The composited stones were scrubbed with a nylon-bristled brush in a bucket with 120 ml of water, and samples were kept on ice in the dark until they were brought back to the laboratory. This material was analysed for chlorophyll *a* (Chl. *a*, 90% acetone, with phaeophytin correction, APHA 1989), DM and AFDM (methods same as for BPOM). The length, breadth, and width of the stones was measured and used to estimate the stones surface area following Dall (1979). The exposed surface area (half of the total estimated surface area) was used to calculate algal biomass.

Ten 0.04 m² Surber samples (250 µm mesh nets) were taken at random points across transects in each reach. Samples were preserved (70% iso-propanol) in the field. In the laboratory, the samples were spread in white trays, and animals were removed, counted, and identified as far as possible following keys in Chapman & Lewis (1976) (Crustacea), Winterbourn (1973) (Mollusca), and Winterbourn & Gregson (1989) (Insecta). The length of each individual invertebrate was measured under a dissecting

microscope, by lining the individual up with a piece of 1 mm grid paper below the petri dish. For abundant taxa, a random sample of 20 individuals was measured. Body length measurements were converted into biomass estimates for each taxonomic group, using length-dry mass regression equations given in Towers et al. (1994) (Insects), Meyer (1989) (Amphipoda, Nematoda, Naididae), Dyer & Quinn (unpubl. data) (*Latia neritoides*, *Potamopyrgus antipodarum*), and Parkyn (unpubl. data) (*Paranephrops planifrons*).

The Quantitative Macroinvertebrate Community Index (QMCI) of Stark (1985) was calculated using the scoring system updated in Stark (1993). This biotic index of organic pollution assigns pollution tolerance scores to individual taxa within a sample, so that the calculated index reflects the level of organic pollution at a site.

The elutriated organic matter from the 10 Surber samples at each site was pooled and the dry mass (70°C to constant mass) of medium and coarse particulate organic matter (MPOM = 0.25–1 mm, CPOM >1 mm material, respectively) measured.

Data analysis

Differences in channel morphology, habitat variables, and community attributes between the sites were compared using one-way ANOVA (Data Desk 6.0) on log₁₀(x+1) transformed count data, and arcsin√x transformed percentage data. ANOVAs giving significant factor effects were followed by a Bonferroni post-hoc test, to identify differences between site means. Sites were classified according to water quality characteristics using cluster analysis in Data Desk 6.0 (calculated from Euclidean distances). Catchment differences (Mangaotama versus Kiripaka) were tested using a student *t*-test on transformed data. Site differences in community composition of the benthic samples were assessed using non-metric multidimensional scaling (NMS) (PCORD, Sorenson distances, random number seeds 1, 100). The associations between the NMS ordination scores and abiotic variables were tested with Spearman rank correlations (Data Desk 6.0).

RESULTS

Stream morphology and water quality

Levels of stream shade were greater at the forested sites than at the pasture sites (Table 2). Two of the transition zone sites (KT1 and MT1) had shade levels approximately half those of the undisturbed

Table 2 Physical characteristics of each site. First letter of the site code is for the stream name K for Kiripaka and M for Mangakowhai, the second letter is for the land use around the study reach (P for pasture, T for transition forest, N for native forest control site). Stream shade relates to the proportion of light reaching the water surface compared to an open reference site, measured in diffuse non-interceptance light (DIFN).

	KP1	KT1	KP2	KT4	KT2	KN1	MPI	MT1	MT2
Distance into forest patch (m) (average for KT2)	—	30–80	—	10–60	270–320	—	—	10–60	260–310
Catchment size (km ²)	0.181	0.206	0.162	0.187	0.453	0.421	0.663	0.825	0.931
Stream order	1	1	1	1	2	1	1	2	2
Stream shade (DIFN)	0.579	0.028	0.203	0.008	0.009	0.011	0.467	0.02	0.013
Channel slope (%)	13.94	7.44	8.42	9.03	5.00	3.46	2.48	1.01	2.62
Pfankuch (1975) instability index	87	128	87	88	96	80	68	81	70
Altitude (m a.s.l.)	180	160	160	160	120	140	80	80	80

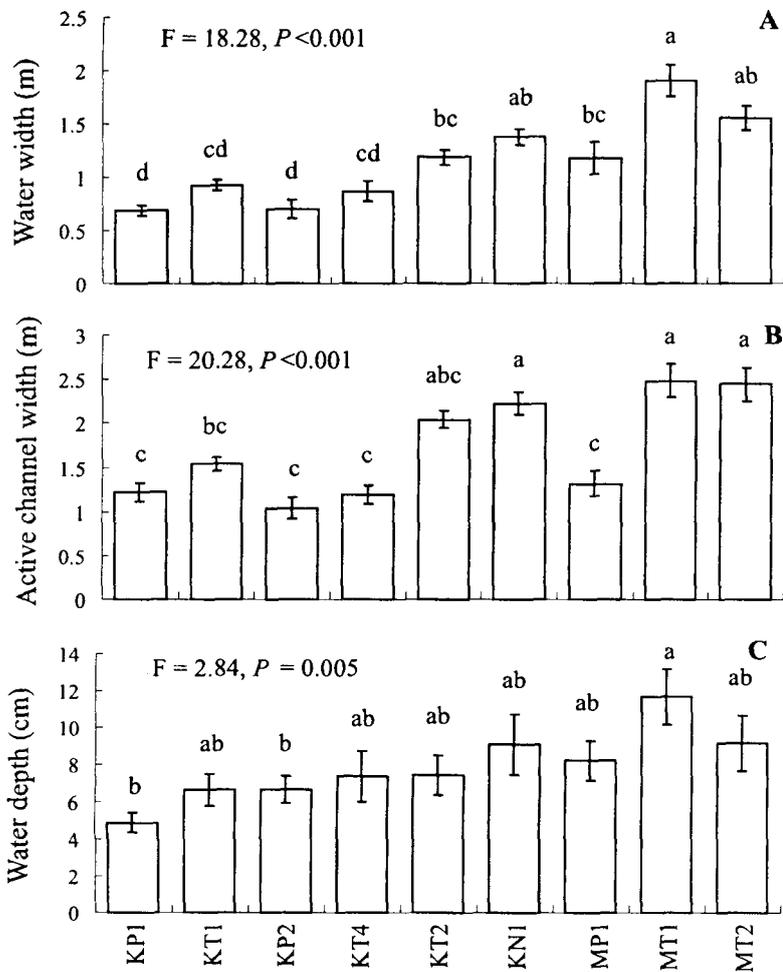
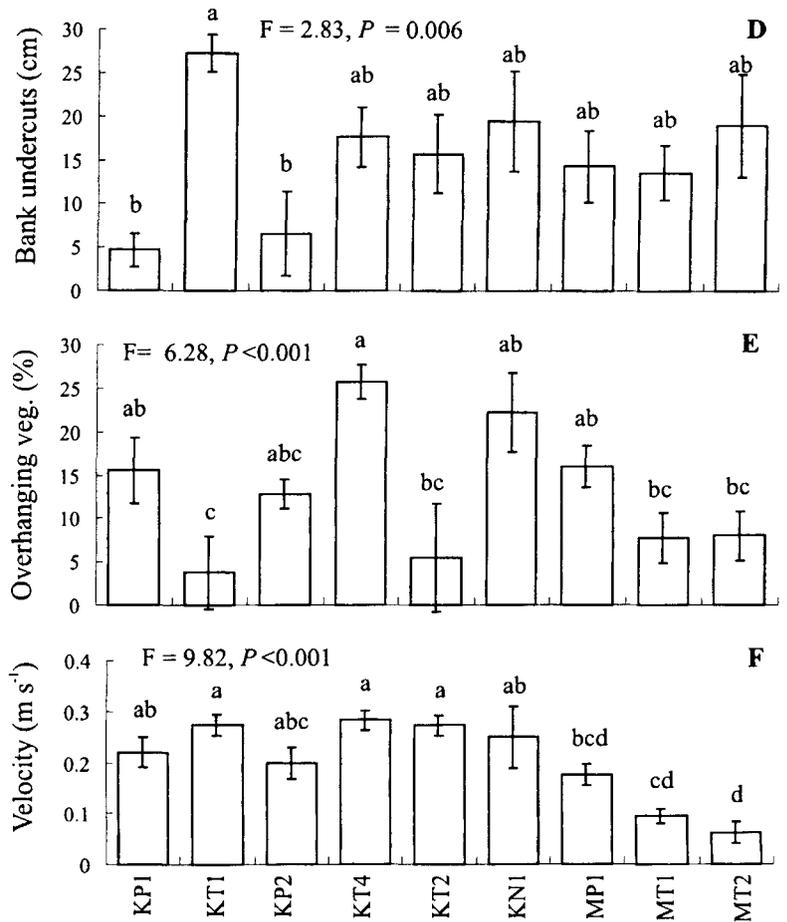


Fig. 2 (and opposite) Stream characteristics (mean \pm 1 SE) amongst sites with differing riparian management. ANOVA (log $x+1$ transformed data) result shown, bars with different letters are significantly different (Bonferroni post-hoc test; $P < 0.05$). Average cross-sectional velocity ($m s^{-1}$) is the mean of 10 cross-sectional values (cross-sectional velocity is the average of three velocities taken across the channel width).

native site (KN1). Channel slope ranged from 1.0% at MT1 to 13.9% at KP1. There was a major difference in slope between KP1 and KT1 (7.4%), but not at the other pasture-transition zone pairings. Slope in the 100 m reach between KP1 and KT1 was 16%. Values for the Pfankuch Index were considerably higher at KT1 than at the other sites (Table 2), reflecting unstable conditions for upper and lower banks and streambed components at this site. KT2 also had a higher score than the pasture sites and native forest reference site.

Both water (range 0.69–1.9 m) and channel width (range 1.04–2.48 m) varied significantly between sites, following a pattern of increasing catchment size ($P < 0.001$) (Fig. 2A,B). Channel widths at the transition zone sites (KT1 = 0.92; KT4 = 0.86; MT1 = 1.9 m) were slightly higher than at the corresponding pasture sites (KP1 = 0.69; KP2 = 0.70; MP1 =

1.17 m), although the difference was only significant ($P < 0.05$) for the MP1–MT1 comparison where a side-stream entered between the sites. Water depth did not vary significantly between pasture-transition zone pairings, although depth did vary between sites ($P = 0.005$; Fig. 2C). The KT1 transition zone site had significantly more bank undercuts (one-way ANOVA, $P = 0.006$; Bonferroni test $P < 0.05$;) than the other sites (Fig. 2D). The two Kiripaka pasture sites had the least bank-undercutting. The percentage of the channel covered by overhanging vegetation varied significantly between sites ($P < 0.001$). KT4 had the greatest amount (26.8%) of overhanging vegetation (Fig. 2E); this was mostly the riparian herb Parataniwha (*Elastostema rugosum*), which was also common at KN1. KT1 had the least overhanging vegetation (3.8%). Mean water velocity (Fig. 2F) varied between sites ($P < 0.001$), with



Mangakowhai sites generally having lower water velocities than Kiripaka sites. Within the Kiripaka catchment there were no significant differences between site pasture-transition zone pairings (Fig. 2D). In the Mangakowhai catchment there was a trend of decreasing velocities down stream into the forest (MP1 = 0.18; MT1 = 0.09; MT2 = 0.06 $m s^{-1}$), but this was not statistically significant ($P > 0.05$).

Results from the one-off sampling of water quality variables at each site are presented in Table 3. At all three transition zone sites immediately inside the forest, there were decreases in pH, and increases in turbidity, DRP, ammonium nitrogen, dissolved inorganic nitrogen, and dissolved organic carbon relative to the upstream pasture site. KT2 and MT2, the two transition zone sites further into the forest, also followed the trend for pH to decrease, and for turbidity and DRP to increase (Table 3).

Cluster analysis of water quality variables (Fig. 3) primarily separated the Mangakowhai from the Kiripaka, which were sampled 6 weeks apart. The Mangakowhai sites had significantly higher electrical conductivity (EC, t -test, $P < 0.001$), dissolved organic carbon (DOC, $P < 0.001$), alkalinity (ALK, $P < 0.001$), and pH ($P = 0.003$) than the Kiripaka sites (Table 3). The Mangakowhai also had significantly lower dissolved reactive phosphorus (DRP, $P < 0.001$), nitrate/nitrite-nitrogen (NO_3/NO_2-N , $P = 0.001$), suspended solids (SS, $P = 0.01$), and dissolved inorganic nitrogen (DIN, $P = 0.005$). The pasture site MP1 separated first from the forested Mangakowhai sites. In the Kiripaka catchment, the native site (KN1) was least like the other sites. The native site had the lowest DIN, NH_4-N and NO_3/NO_2-N (Table 3). The native site also had low SS and turbidity, and high levels of ALK and DRP. KT2

which was 300 m into the forest was the next closest in water quality to the native site. The pasture-transition pairs of KT1–KP1 and KP2–KT4 grouped together, with the pairs splitting off at the final division. The pasture sites generally had higher pH, but lower DRP, DOC, ALK, EC, and turbidity compared with the paired transition zone sites (Table 3).

Benthic habitat and food resources

The size distribution of streambed sediments showed major differences between the nine sites (Fig. 4). The proportion of fine sediment (silt/sand) was highest at KT1 (37%), with levels being higher than at the upstream pasture site (KP1; 21%) and the site 300 m further into the forest (KT2; 17%). The same pattern of initial increase followed by a decrease further into native forest was also evident at the Mangakowhai sites (MP1 = 17; MT1 = 26; MT2 = 11%). The native reference site (KN1) had the lowest proportion of fines (5%). Bedrock was a major constituent of the bed at many of the sites (maximum at KP1; 47%), with the exception of KT1 (0.4%). Woody debris made up little of the streambed habitat (range 0–5.6%), but was higher in forested sites.

The concentrations of SIS and BPOM varied significantly between the sites ($P < 0.001$ for both), and were highest at the Mangakowhai sites (MP1; MT1; MT2; Fig. 5A,B). In the Kiripaka, SIS and BPOM were lowest in the native forest site (KN1; 105 and 10 g m⁻², respectively) and highest at KT1 (2259 and 171 g m⁻² respectively). Levels of SIS decreased significantly ($P < 0.05$) from KT1 to KT2 (571 g m⁻²). SIS and BPOM did not vary significantly between the pasture-transition paired sites. Levels of FPOM and CPOM, collected in benthic Surber

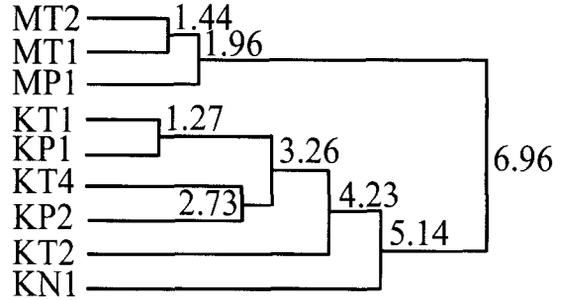


Fig. 3 Cluster analysis of 10 water quality variables (pH, electrical conductivity (EC), turbidity (Turb.), suspended solids (SS), DRP, NH₄-N, NO₃/NO₂-N, dissolved inorganic nitrogen (DIN), dissolved organic carbon (DOC), alkalinity (ALK)) from nine sites varying in riparian vegetation. Euclidean distance is given at each split.

samples, were highly variable between sites (FPOM range 6.4–39.0 g m⁻²; CPOM 4.5–104.0 g m⁻²) and were not significantly different ($P > 0.05$ for both; Fig. 5C,D). Chlorophyll *a* and AFDM levels of benthic periphyton were highest in the pasture sites (Fig. 6A,B), but a comparison of pasture (Chl. *a* range 15.6–156.4 mg m⁻²; AFDM 8.6–21.8 g m⁻²) and forest sites (Chl. *a* range 1.1–18.0 mg m⁻²; AFDM 1.3–4.8 g m⁻²) gave non-significant results (t -test, $P > 0.05$). However, there was a trend of decreasing chlorophyll *a* and AFDM levels with distance into the native forest.

Benthic macroinvertebrates

There were statistically significant differences between invertebrate densities at the nine sites ($P < 0.001$; Fig. 7A; Appendix 1). Invertebrate density

Table 3 Values for water quality variables at sites varying in riparian vegetation (mean of two 1-litre samples taken from the top and the bottom of each 50 m reach). (EC=electrical conductivity; Turb.=turbidity (NTU=nephelometric turbidity units); SS=suspended solids; DRP = dissolved reactive phosphorus; DIN=dissolved inorganic nitrogen; DOC=dissolved organic carbon; ALK=alkalinity (CaCO₃.)

Site	pH	EC (µS cm ⁻¹)	Turb. (NTU)	SS (mg litre ⁻¹)	DRP (ppb)	NH ₄ -N (ppb)	NO ₃ /NO ₂ -N (ppb)	DIN (ppb)	DOC (ppm)	ALK (ppm)
KP1	8.05	80.05	3.96	12.15	33	11	1433	1444	0.92	14.5
KT1	7.93	79.2	4.86	12.5	23.5	12	1394	1406	0.74	14.3
KP2	7.98	87.3	4.89	15.35	67.5	11	2611	2621	1.17	15
KT4	7.90	87.75	5.5	8.65	66	6	2332	2338	1.11	16.8
KT2	7.87	88.95	13.48	28.95	39.5	6	1581	1586	1.24	17
KN1	7.93	104.85	2.68	4	54	4	164	168	0.61	27
MP1	8.31	240.5	1.25	1.35	2	9	339	348	2.13	96
MT1	8.21	222	2.37	10.3	3.5	11	274	285	2.18	85
MT2	8.09	218	4.12	1.8	4.5	11	316	327	2.04	83

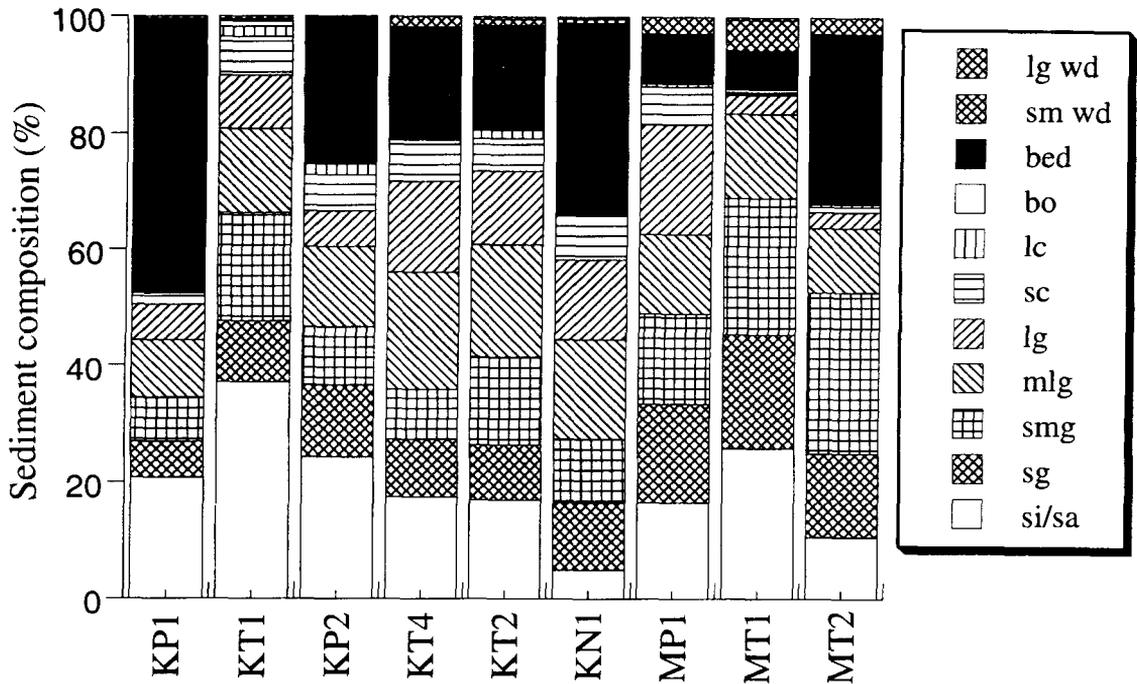


Fig. 4 Stream sediment particle size distribution (as %) of sites varying in riparian vegetation and distance into forest. From c. 100 observations (si/sa=silt/sand (<2 mm); sg=small gravel (2–8 mm); smg = small-medium gravel (8–16 mm); mlg=medium-large gravel (16–32 mm); lg=large gravel (32–64 mm); sc=small cobble (64–128 mm); lc=large cobble (128–256 mm); bo=boulder (>256 mm); bed=bedrock; sm wd=small wood (diameter<10 cm); lg wd=large wood (diameter>10 cm)).

was greater at KP1 (9963 m⁻²) and MP1 (8817 m⁻²) than at other sites (range 1110–3947 m⁻²; $P < 0.05$). The pasture sites were dominated by Diptera taxa (KP1 = 85%; KP2 = 76%; MP1 = 42%), with a smaller proportion of Trichoptera (15%) and Mollusca (8%) at MP1. The forested sites were dominated by taxa belonging to the orders Ephemeroptera, Plecoptera, and Trichoptera (EPT). In all instances the pasture sites had significantly higher densities of invertebrates than their paired transition site (Appendix 1). Invertebrate density did not vary significantly amongst the forested sites.

The mean number of taxa found per Surber sample (0.04 m²) varied significantly between sites ($P < 0.001$), with MP1 having a greater mean number of taxa (21) than KT2 (8.5) or KN1 (9.6) (Fig. 7B). There were no significant differences between forested sites, or between pasture-transition site pairings.

There were statistically significant differences in biomass between the sites ($P = 0.003$; Fig. 7C). Invertebrate biomass was highest at MP1 (1101 mg m⁻²), lower at KP1 (440 mg m⁻²), and lowest at KT1

(66 mg m⁻²). Invertebrate biomass in the pasture sites was dominated by Diptera taxa, mainly Chironomidae. Invertebrate biomass was high in the “other” category of MP1 (286 mg m⁻²) because of the presence of several large Oligochaeta. Invertebrate biomass was significantly higher in the pasture site in two out of the three transition pairs (Fig. 7C).

Percentage of EPT individuals gave a contrasting pattern to that of total invertebrate density (Fig. 7D). There were significant differences between sites ($P < 0.001$), with forested sites having generally higher percentages of EPT individuals (range 35–76%) than pasture sites (14–25%). However, the difference was statistically significant at only one pasture-transition pairing (KP2–KT4).

Values for the QMCI, which also varied significantly between sites ($P < 0.001$), reflected the same pattern as the EPT results, with the three pasture sites and the physically unstable transition site, KT1, significantly lower than the other sites (Fig. 7E). QMCI values at the forested sites, other than KT1, were not significantly different.

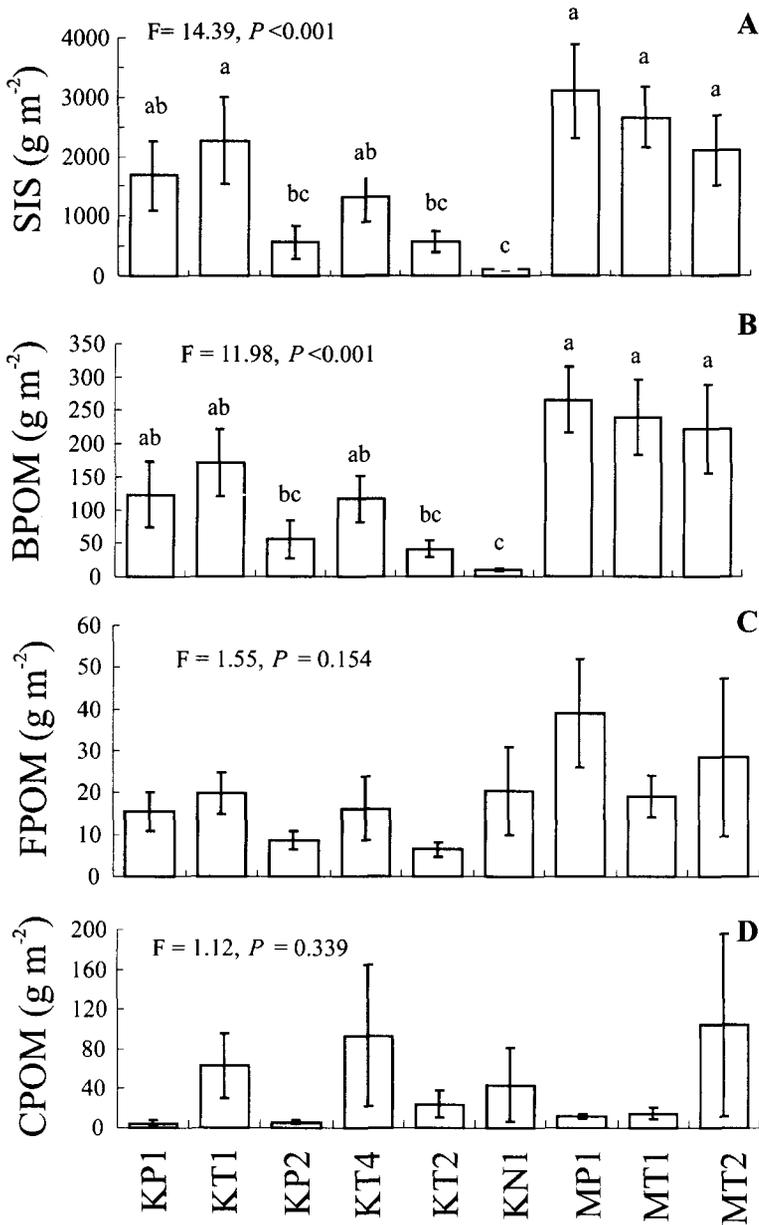


Fig. 5 Riparian vegetation effects on benthic habitat characteristics (mean \pm 1 SE, ANOVA results shown, bars with different letters are significantly different with Bonferroni post-hoc test). **A** and **B**, suspended inorganic sediments (SIS) and benthic particulate organic matter (BPOM) respectively, average of seven benthic samples; **C** and **D**, mean mass of fine and coarse particulate organic matter from 10 Surber samples.

The NMS ordination of invertebrate community composition at each site yielded a 2-dimensional solution with relatively low stress ($S = 23.7$; Fig. 8). The native forest samples had low scores on axis 1 and high scores on axis 2, whereas the pasture samples are high on axis 1, but low on axis 2. The Mangakowhai samples tended to have low axis 1 and axis 2 scores. All three early transition zone sites

(KT1, KT4, and MT1) showed a strong shift in community composition away from that of the upstream pasture sites and towards the native forest reference site. The two late transition zone sites (KT2 and MT2) showed a further shift towards the native forest reference site.

There were significant negative correlations between alkalinity, EC, and velocity and the first axis

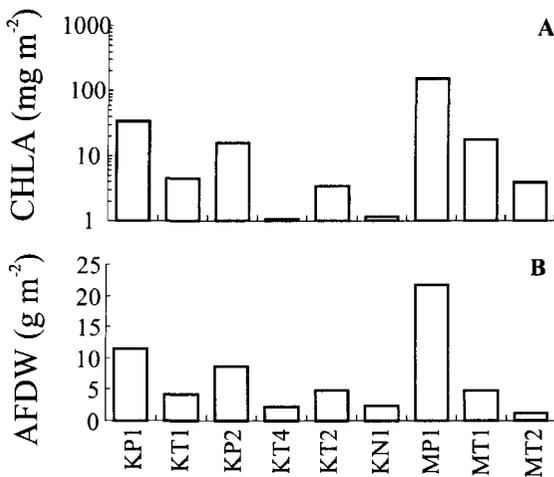


Fig. 6 Chlorophyll *a* and ash free dry mass (AFDM) densities (composite sample of seven stones) from nine sites varying in riparian vegetation.

of the benthic invertebrate NMS analysis (Spearman rank correlation, $P < 0.05$) (Table 4). Turbidity, DRP, and velocity were positively correlated and chlorophyll *a* and pH negatively correlated with axis 2 of the NMS analysis ($P < 0.001$).

DISCUSSION

In this study we set out to test predictions (cf. Table 1) regarding the restoration of forest stream characteristics as streams flowed from pasture into native forest remnants at sharp transition zones. We considered such a situation to be analogous to the discontinuous restoration of stream riparian zones, as may occur where patterns of land ownership and economic considerations may preclude more extensive restoration schemes. Central to the study was the assumption that physical, chemical, and biological characteristics of pasture streams would be restored to near native forest conditions given a sufficient length of native forest riparian zone. However, as noted by Storey & Cowley (1997), it was also expected that the distance of forest required for restoration of "native" conditions would vary with the variable of interest.

As predicted, the high levels of shade characteristic of New Zealand forest streams (>95%; Davies-Colley & Payne 1998) were restored to the study streams immediately below the transition from pasture into forest. Channel width also followed the prediction of widening immediately below the forest

margin to become similar to the native forest condition several hundred metres into the forest. The narrowing of stream channels under pastoral land use is a well known phenomenon (Sweeney 1993; Davies-Colley 1997; Trimble 1997). The high levels of light reaching pasture stream channels allow grasses to colonise exposed areas of streambed and trap and consolidate alluvium and colluvium leading to a narrowing of the active channel (Davies-Colley 1997). It has been suggested (Davies-Colley 1997; Trimble 1997) that re-establishment of riparian canopy could cause stream channels to widen as a result of the loss of sediment stabilising ground cover. This channel adjustment may result in increased sediment yields for a period of years with potential adverse effects on stream ecosystems.

Quinn et al. (1997) found that in addition to narrower channels, pasture streams in and adjacent to Whatawhata Research Centre also had greater average current velocity than forest streams (water depth did not vary significantly). Hence we predicted that following the pattern of channel widening of streams at the transition from pasture to native forest, velocity would also decrease with distance into the forest. However, although a trend of decreasing current velocity with distance into the forest remnant was observed at the Mangakowhai sites, this pattern was not seen in the Kiripaka site, where stream discharge was greater at the site furthest into the forest.

A number of studies have shown that agricultural land use often results in an increased supply of fine sediment to streams (Walling 1990; Richards et al. 1993). Presumably as a result of this increased supply, Quinn et al. (1997) found increased amounts of fine sediment in the benthic zone of pasture streams at Whatawhata. In the present study it was predicted that bed sedimentation levels would decrease with distance into native forest remnants as sediment supply decreased, to be balanced by downstream export and deposition in storage zones (Einstein 1968). The percentage of fine sediments (< 2 mm) in the streambed was highest at two of the early transition zone sites (KT1 and MT1), and lowest at the native forest reference site. In both instances, though, fine sediment levels decreased at the site further (c. 300 m) into the forest, although levels were still considerably higher than those at the native forest site. In this study we only measured levels of surficial fine sediment. It is well known that fine sediments filter down into the streambed (Einstein 1968) and often form an impermeable layer (Wood & Armitage 1997). This process of colmation can sever the ecologically important links between the stream benthic

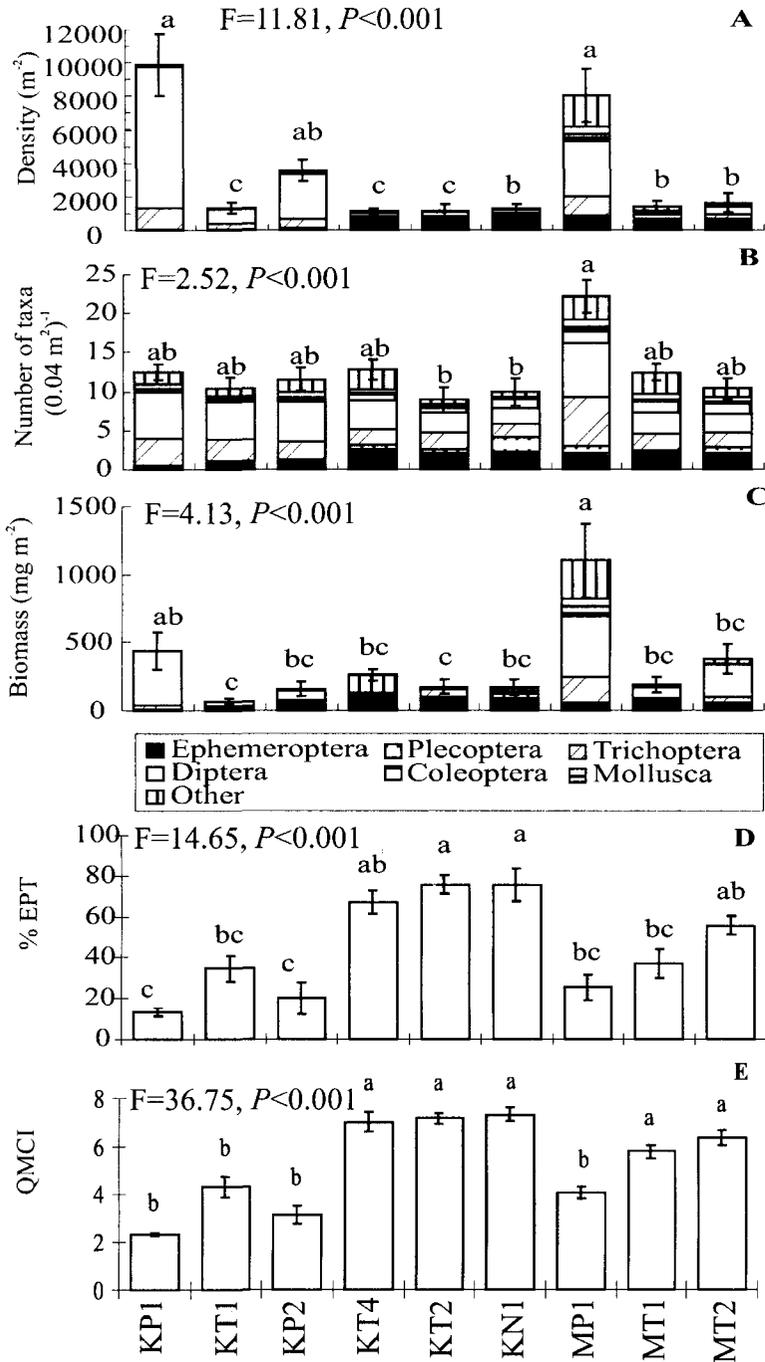


Fig. 7 Riparian management effects on: **A**, macroinvertebrate abundance; **B**, invertebrate community richness; **C**, invertebrate biomass; **D** percentage of individuals in the Ephemeroptera, Plecoptera, and Trichoptera (%EPT); and **E**, Quantitative Macroinvertebrate Community Index (QMCI) value (mean of 10 Surbers \pm 1 standard error, ANOVA results shown, bars with different letters are significantly different in Bonferroni post-hoc test ($P < 0.05$)).

and hyporheic zones (Brunke & Gonser 1997). Indeed, Boulton et al. (1997) suggested that the depauperate fauna of the hyporheic zone in pasture streams at Whatawhata may be due to reduced

interstitial hydraulic conductivity brought about by colmation in these streams.

Maintenance of an elevated sediment supply from pasture headwaters may limit the potential for

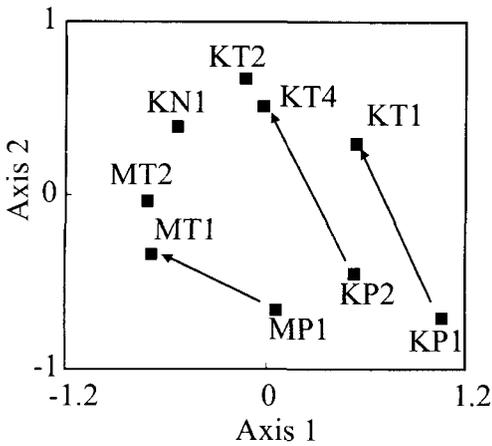


Fig. 8 Non-metric multidimensional scaling 2-dimensional ordination plot of 90 benthic invertebrate samples collected from nine sites differing in riparian vegetation. Only the average axis scores of the samples from each site are shown; arrows show the direction of change between pasture sites and the first corresponding transition zone.

restoration of streambed substrate composition regardless of the distance of riparian restoration in areas further down the catchment. Suggestions by Davies-Colley (1997) that riparian restoration could result in short-term increases in bank erosion would only exacerbate this problem. More research is required to determine the temporal and spatial dynamics of fine sediment deposition and its hydraulic, chemical, and biological consequences, both in the short-term and longer-term in pasture streams.

Cluster analysis showed that the native forest reference site (KN1) had a different water chemistry to all other sites. Furthermore, two of the sites immediately below pasture-forest transition zones (KT1 and KT4) were more similar in water chemistry to their pasture stream sources than to the site further into the forest. This suggests that, as predicted, water chemistry does not change rapidly at the pasture-forest transition, and even at distances of c. 300 m into forest remnants water chemistry was still markedly different from that of the native forest stream condition. Storey & Cowley (1997) found nitrate, nitrite, SS, and phosphate produced variable results in their three transition zones over time, but

Table 4 Results of Spearman rank correlations between mean habitat variables and mean biological variables for the nine sites, including axis 1 and axis 2 of the Non-metric Multidimensional Scaling (NMS) analysis. $N=9$, significant ($P < 0.05$) correlation coefficients are given in bold. (% EPT = % of individuals belonging to Ephemeroptera, Plecoptera, and Trichoptera; QMCI = Quantitative Macroinvertebrate Community Index; AFDM = ash free dry mass; SS = suspended sediments; DRP = dissolved reactive phosphorus; DIN = dissolved inorganic nitrogen; DOC = dissolved organic carbon; EC = electrical conductivity; Chl. a = chlorophyll a ; DIFN = diffuse non-interceptance; BPOM = benthic particulate organic matter; SIS = suspendable inorganic sediment.)

Variable	Taxa richness	Total abundance	% EPT	QMCI	NMS axis 1	NMS axis 2
AFDM	0.55	0.633	-0.667	-0.667	0.60	-0.633
Turbidity	-0.783	-0.85	0.667	0.667	-0.233	0.917
SS	-0.533	-0.45	0.117	0.117	0.40	0.533
DRP	-0.583	-0.733	0.55	0.55	0.05	0.717
NH ₄ -N	0.101	0.034	-0.151	-0.151	-0.118	-0.059
NO ₃ /NO ₂ -N	-0.017	-0.233	-0.15	-0.15	0.483	0.233
DIN	-0.067	-0.35	-0.167	-0.167	0.467	0.283
DOC	0.15	-0.15	0.283	0.283	-0.65	0.133
Alkalinity	0.133	-0.067	0.383	0.383	-0.717	0.067
EC	0.133	-0.067	0.383	0.383	-0.717	0.067
Chl. a	0.733	0.883	-0.833	-0.833	0.40	-0.90
Light (DIFN)	0.40	0.717	-0.55	-0.55	0.30	-0.633
BPOM	0.60	0.50	-0.467	-0.467	-0.05	-0.55
SIS	0.467	0.383	-0.40	-0.4	0.017	-0.433
Pfankuch	-0.452	-0.485	0.042	0.042	0.452	0.469
pH	0.745	0.778	-0.544	-0.544	-0.109	-0.812
Velocity	-0.6	-0.667	0.383	0.383	-0.700	0.883
% silt and sand	0.267	0.10	-0.533	-0.533	0.50	-2.17

there was some evidence of significant processing over 600 m. In the present study SS levels showed no trend with extent of riparian forest remnant. Nitrogen ($\text{NH}_4\text{-N}$, $\text{NO}_3\text{-N}$, DIN) concentrations were lowest in the native site, and there was no evidence of significant processing over the 300 m between the pasture-forest transition and the site furthest into the forest in either catchment (Kiripaka or Mangakowhai). Cooper & Thomsen (1988) also failed to find significant nitrate removal from a stream flowing through native forest, which they attributed to low levels of autochthonous production. The lack of significant processing of DIN at distances of c. 300 m in both catchments, and Kiripaka in particular, suggest that if reduced nitrogen loads are a goal for stream restoration then discontinuous riparian planting may not be effective without additional targeting of nutrient sources (Quinn et al. 1993).

Levels of dissolved reactive phosphorus were higher in the forested sites than the pasture sites in this study, and there was a trend for increasing DRP levels with distance into the forest remnant in the Mangakowhai catchment. In two of the three streams studied by Storey & Cowley (1997), phosphate levels were significantly greater at sites 350–450 m into forest remnants compared with their upstream pasture site. These results suggest in-stream processing of DRP is lower in the forested sections than in pasture, where periphyton and macrophytes remove a major proportion of the available nutrients.

Epilithon biomass was greater in the pasture sites than at sites in the forest. However, there were appreciable levels of chlorophyll *a* at two of the three sites 50 m into the forest, despite the high levels of shade. In the Mangakowhai catchment, periphyton biomass was extremely high at the pasture site (156 mg m^{-2}), was reduced 9-fold at the site immediately inside the forest (98 % shade) and was reduced a further 5-fold, down to 4 mg m^{-2} , at a distance of c. 300 m into the forest. These results suggest that sloughed periphyton material from upstream pasture sites drifts down stream and is deposited in the forested sites, where it may be broken down, contributing to particulate carbon stores.

From the results of Quinn et al. (1997) it was hypothesised that CPOM biomass would be greatest in the native sites, intermediate in the transition sites and lowest in the pasture sites. The results of the benthic POM analysis were very variable and showed no significant trend with the presence of riparian forest, although mean CPOM levels (measured as dry mass only) were higher in the forested sites than those in the pasture sites. The method of

collection of benthic particulate organic matter did not discriminate between periphyton and allochthonous material and this may have led to an overestimation of benthic organic matter in the pasture streams.

Land use has marked influences on benthic invertebrate community composition in New Zealand streams (Harding & Winterbourn 1995; Quinn et al. 1997; Townsend et al. 1997). Quinn et al. (1997) found that invertebrate density was 3-fold higher in pasture than in native forest streams. Biomass also tended to be greater in pasture streams. These differences were mainly because of the high numbers of chironomids and the snail *Potamopyrgus antipodarum* (Gray). In contrast, the density of individuals belonging to the Ephemeroptera, Plecoptera, and Trichoptera, groups generally considered to be sensitive to organic enrichment, were greater in the native forest than in the pasture streams. Ordination of invertebrate assemblages at the nine sites supported our predictions of strong shifts in community composition at the transition zone sites, and further shifts towards the native forest condition at the sites furthest into the forest remnant.

Water quality variables correlated strongly with both axes of the invertebrate community ordination, suggesting that water quality was an important factor controlling the observed differences in invertebrate community composition. Quinn et al. (1997) also found significant correlations between the axes of a community ordination (DECORANA) and several water quality variables, although in their analysis shade gave the strongest correlation.

Invertebrate density and biomass was highest in the pasture sites, especially KP1 and MP1. The prediction that the transition zone sites would have the lowest density, biomass, and species richness was not supported by our results. Nor did species richness follow the trend predicted in Table 1. Two large-scale studies of land-use effects in streams (Quinn et al. 1997; Townsend et al. 1997) both failed to find significant differences in species richness across land-use types, but as in our study, both found major differences in the composition of invertebrate communities. In the present study Diptera species, predominantly Chironomidae, dominated pasture sites, whereas Ephemeroptera, Trichoptera, and Plecoptera species dominated forested communities. As a reflection of the differences in the numbers of "sensitive" EPT individuals, QMCI values were also significantly higher in the forested sites than the pasture site, as predicted. There was an exception, with KT1 having a lower QMCI than the other

forested sites. Similarly, Storey & Cowley (1997) found that MCI values were higher in sites in remnant forest, although a silt-affected forest remnant site had a MCI value equivalent to the pasture sites.

The transition site KT1 had unstable undercut banks, high levels of interstitial fine sediment, and a QMCI value significantly lower than the other forested sites. The other two sites immediately inside forest remnants showed an improvement in habitat conditions compared with the upstream pasture sites. The land-use history of KT1 was the same as KT4 and the forest cover was of a similar density and species composition (J. Halliday pers. obs.). KT1 and KT4 also had similar aspect, catchment size, and geology. Stock could get under the fence at KT1 unlike KT4, although there was little evidence they followed the stream channel for more than 30 m, where the forest canopy closed over and the study reach began. The major difference between KT1 and the other transition sites was channel slope. The pasture stream above KT1 had a very steep catchment (land slopes of 10–40%) with a large, still active slip in the headwaters. A major break in channel slope occurred at the fenceline separating the pasture from the forest remnant. The combination of high sediment supply to the stream, the change in slope below the fenceline and the lack of sediment trapping devices in the stream channel (e.g., pasture grasses and macrophytes) inside the forest remnant could account for the observed poor habitat quality and invertebrate community composition at KT1.

If this explanation is correct a note of warning regarding the discontinuous restoration of riparian forest cover is warranted. Davies-Colley (1997) and Trimble (1997) have warned that restoration of riparian canopy in pasture streams may carry a risk of increased bank erosion as sediment deposits, previously stabilised by encroaching pasture grasses and aquatic macrophytes, become destabilised with consequences for downstream water quality and biota. The results of the present study, with respect to observations at KT1, suggest that in determining where riparian restoration should be focused, the topography of the catchment and the longitudinal profile of the stream should be taken into account. Stream reaches of low slope, which may be acting as sediment storage zones, may require careful planning in terms of riparian restoration. Whereas steeper sections, where sediment trapping capabilities are lower, could have riparian canopy cover restored with potentially little risk of bank erosion and downstream sediment problems.

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REFERENCES

- APHA 1989: Standard methods for the examination of water and wastewater. 17th edition. Washington, DC, American Public Health Association, American Water Works Association, Water Pollution Control Federation.
- Boulton, A. J.; Scarsbrook, M. R.; Quinn, J. M.; Burrell, G. P. 1997: Land-use effects on the hyporheic ecology of five small streams near Hamilton, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 31: 609–622.
- Brunke, M.; Gonsler, T. 1997: The ecological significance of exchange processes between rivers and groundwater. *Freshwater Biology* 37: 1–33.
- Burton, T. M.; Likens, G. E. 1973: The effect of strip-cutting on stream temperature in the Hubbard Brook Experimental Forest, New Hampshire. *Bio-science* 23: 433–435.
- Chapman, M. A.; Lewis, M. H. 1976: An introduction to the freshwater Crustacea of New Zealand. Auckland, Collins.
- Collier, K. J. 1992: Assessing river stability: use of the Pfankuch Method. Science and Research Internal Report. Department of Conservation, Wellington, New Zealand. 17 p.
- Collier, K. J.; Cooper, A. B.; Davies-Colley, R. J.; Rutherford, J. C.; Smith, C. M.; Williamson, R. B. 1995: Managing riparian zones: a contribution to protecting New Zealand's rivers and streams. Volume 2: Guidelines. Wellington, New Zealand, Department of Conservation.
- Cooper, A. B.; Thomsen, C. E. 1988: Nitrogen and phosphorus in streamwaters from adjacent pasture, pine and native forest catchments *New Zealand Journal of Marine and Freshwater Research* 22: 279–291.
- Dall, P. C. 1979: A sampling technique for littoral stone-dwelling organisms. *Oikos* 33: 106–112.
- Davies-Colley, R. J. 1997: Stream channels are narrower in pasture than in forest. *New Zealand Journal of Marine and Freshwater Research* 31: 599–608.

- Davies-Colley, R. J.; Payne, G. W. 1998: Measuring stream shade. *Journal of the North American Benthological Society* 17: 250–260.
- Einstein, H. A. 1968: Deposition of suspended particles in a gravel bed. *Journal of the Hydraulics Division of the American Society of Civil Engineers* 94: 1197–1205.
- Gregory, S. V.; Swanson, F. J.; McKee, W. A.; Cummins, K. W. 1991: An ecosystem perspective of riparian zones. *Bioscience* 41: 540–551.
- Harding, J. S.; Winterbourn, M. J. 1995: Effects of contrasting land use on physico-chemical conditions and benthic assemblages of streams in Canterbury (South Island, New Zealand) river systems. *New Zealand Journal of Marine and Freshwater Research* 29: 479–492.
- Maasdam, R.; Smith, D. G. 1994: New Zealand's National River Water Quality Network 2. Relationships between physico-chemical data and environmental factors. *New Zealand Journal of Marine and Freshwater Research* 28: 37–54.
- Meyer, E. 1989: The relationship between body length parameters and dry mass in running water invertebrates. *Archiv für Hydrobiologie* 2: 191–203.
- Pfankuch, D. J. 1975: Stream reach inventory and channel stability evaluation. U.S.D.A. Forest Service Report, Region 1, Missoula, Montana, United States.
- Quinn, J. M.; Cooper, A. B. 1997: Land-water interactions at Whatawhata, New Zealand: introduction and synthesis. *New Zealand Journal of Marine and Freshwater Research* 31: 569–577.
- Quinn, J. M.; Cooper, A. B.; Davies-Colley, R. J.; Rutherford, J. C.; Williamson, R. B. 1997: Land-use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research* 31: 579–597.
- Quinn, J. M.; Cooper, A. B.; Williamson, R. B. 1993: Riparian zones as buffer strips: a New Zealand perspective. In: Bunn, S. E.; Pusey, B. J.; Price, P. ed. Ecology and management of riparian zones in Australia. Proceedings of a National Workshop on research and management needs for riparian zones in Australia, held in association with the 32nd Annual Congress of the Australian Society for Limnology, Marcoola. Pp. 53–58.
- Quinn, J. M.; Steele, G. L.; Hickey, C. W.; Vickers, M. L. 1994: Upper thermal tolerances of twelve New Zealand stream invertebrate species. *New Zealand Journal of Marine and Freshwater Research* 28: 391–397.
- Richards, C.; Host, G. H.; Arthur, J. W. 1993: Identification of predominant environmental factors structuring stream macroinvertebrate communities within a large agricultural catchment. *Freshwater Biology* 29: 285–294.
- Rutherford, J. C.; Blackett, S.; Blackett, C.; Saito, L.; Davies-Colley, R. J. 1997: Predicting the effects of shade on water temperature in small streams. *New Zealand Journal of Marine and Freshwater Research* 31: 707–721.
- Smith, C. M. 1993: Perceived riverine problems in New Zealand, impediments to environmentally sound riparian zone management, and the information needs of managers. *Water Quality Centre Publication* 24. Hamilton, NIWA. 44 p.
- Stark, J. D. 1985: A Macroinvertebrate Community Index of Water Quality for stony streams. *Water and Soil Miscellaneous Publication No. 87*. Wellington New Zealand, Ministry of Works and Development.
- Stark, J. D. 1993: Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research* 27: 463–478.
- Storey, R. G.; Cowley, D. R. 1997: Recovery of three New Zealand rural streams as they pass through native forest remnants. *Hydrobiologia* 353: 63–76.
- Sweeney, B. W. 1993: Effects of streamside vegetation on macroinvertebrate communities of White Clay Creek in eastern North America. *Proceedings of the Academy of Natural Sciences of Philadelphia* 144: 291–340.
- Towers, D. J.; Henderson, I. M.; Veltman, C. J. 1994: Predicting the dry weight of New Zealand aquatic macroinvertebrates from linear dimensions. *New Zealand Journal of Marine and Freshwater Research* 20: 139–166.
- Townsend, C. R.; Arbuckle, C. J.; Crowl, T. A.; Scarsbrook, M. R. 1997: The relationship between land-use and physicochemistry, food resources and macroinvertebrate communities in tributaries of the Taieri River, New Zealand: a hierarchically scaled approach. *Freshwater Biology* 37: 177–191.
- Trimble, S. W. 1997: Stream channel erosion and change resulting from riparian forests. *Geology* 25: 467–469.
- Upton, S. 1994: Opening address. Implementing the Resource Management Act: Sixth National water Conference, Hamilton, 29–31 August 1994. Pp. 2–8.
- Walling, D. E. 1990: Linking the field to the river: sediment delivery from agricultural land. In: Boardman, J.; Foster, I. D. I.; Dearing, J. A. ed. Soil erosion on agricultural land. Chichester, John Wiley.

Welles, J. M.; Norman, J. M. 1991: Instrument for indirect measurement of canopy architecture. *Agronomy Journal* 83: 818–825.

Winterbourn, M. J. 1973: A guide to the freshwater Mollusca of New Zealand. *Tuatara* 20: 141–159.

Winterbourn, M. J.; Gregson, K. L. D. 1989: Guide to the aquatic insects of New Zealand. *Bulletin of the Entomological Society of New Zealand* 9: 95.

Wood, P. J.; Armitage, P. D. 1997: Biological effects of fine sediment in the lotic environment. *Environmental Management* 21: 203–217.

Appendix 1 Relative abundance (%) of aquatic invertebrate taxa collected in samples from nine stream sites (cf. Fig. 1 for site locations). Mean and standard deviation (SD) for total invertebrate density (numbers m⁻²) are also given. (* = taxa with relative abundance < 1 %; – = not present.)

Taxa	KP1	KT1	KP2	KT4	KT2	KN1	MP1	MT1	MT2
Megaloptera									
<i>Archichauliodes diversus</i> (Walker)	–	–	–	–	2	1	1	1	1
Ephemeroptera									
<i>Acanthophlebia cruentata</i> (Hudson)	–	–	*	*	–	2	–	1	1
<i>Ameletopsis perscitus</i> (Eaton)	–	–	–	–	–	1	–	–	–
<i>Austroclima sepia</i> (Phillips)	–	–	–	–	–	1	*	1	2
<i>Deleatidium</i> spp.	*	3	2	17	44	57	7	21	25
<i>Coloburiscus humeralis</i> (Walker)	–	–	*	1	5	2	1	–	2
<i>Ichthyotus hudsoni</i> (McLachlan)	–	–	–	–	*	*	*	–	–
<i>Mauilulus luma</i> Towns & Peters	–	–	–	–	–	*	–	1	1
<i>Neozephlebia scita</i> (Walker)	–	–	–	–	–	*	–	–	1
<i>Nesameletus</i> sp.	–	–	–	–	1	–	–	*	–
<i>Zephlebia borealis</i> (Phillips)	–	–	–	*	–	–	–	1	1
<i>Zephlebia dentata</i> (Eaton)	*	1	1	37	5	1	1	7	3
<i>Zephlebia inconspicua</i> Towns	–	–	–	–	–	–	–	2	*
<i>Zephlebia spectabilis</i> Towns	–	–	*	–	–	2	–	–	–
<i>Zephlebia versicolor</i> (Eaton)	–	*	–	–	–	–	–	1	–
<i>Zephlebia</i> sp.	*	–	*	–	–	–	*	*	1
Plecoptera									
<i>Austroperla cyrene</i> (Newman)	–	*	–	*	–	1	–	–	*
<i>Megaleptoperla diminuta</i> Kimmins	–	–	–	–	–	–	2	–	–
<i>Spaniocerca</i> sp.	–	–	–	–	*	–	–	–	–
<i>Spaniocerca zelandica</i> Tillyard	*	1	–	1	–	*	–	–	–
<i>Zelandobius confusus</i> (Hare)	–	–	–	*	–	2	–	–	–
<i>Stenoperla prasina</i> (Newman)	–	–	–	–	*	2	–	–	–
<i>Zelandobius furcillatus</i> Tillyard	–	–	–	–	–	1	*	2	1
<i>Zelandoperla fenestrata</i> Tillyard	–	–	–	–	–	*	–	–	–
Trichoptera									
<i>Aoteapsyche catherinae</i> (McFarlane)	–	–	–	4	–	–	–	–	5
<i>Aoteapsyche colonica</i> (McLachlan)	–	–	*	–	–	–	5	–	*
<i>Aoteapsyche rarururu</i> (McFarlane)	2	–	–	–	–	–	–	–	–
<i>Aoteapsyche</i> sp.	2	3	1	–	–	–	*	–	*
<i>Aoteapsyche tepoka</i> Mosely	*	–	–	–	–	–	3	–	–
<i>Costachorema</i> spp.	–	–	*	–	–	–	*	–	–
<i>Helicopsyche</i> sp.	–	–	–	–	–	*	–	–	*
<i>Hudsonema amabilis</i> (McLachlan)	–	–	–	–	–	–	*	–	–
<i>Hydrobiosella mixta</i> (Cowley)	–	–	–	1	–	*	–	–	–
<i>Hydrobiosis</i> sp.	2	4	–	1	1	–	2	–	–
<i>Hydrobiosis</i> sp. C	–	–	–	–	–	–	*	–	–
<i>Hydrobiosis copis</i> McFarlane	*	–	–	–	–	–	–	–	–
<i>Hydrobiosis gollanis</i> Mosely	–	–	–	–	*	–	–	–	–
<i>Hydrobiosis parumbripennis</i> McFarlane	1	10	6	–	1	–	2	–	1
<i>Hydrobiosis umbripennis</i> (McLachlan)	–	–	2	–	–	–	*	–	–

(continued on next page)

Appendix 1 (continued)

Taxa	KP1	KT1	KP2	KT4	KT2	KN1	MP1	MT1	MT2
<i>Neurochorema confusum</i> (McLachlan)	*	6	1	5	2	1	1	1	2
<i>Neurochorema</i> sp.	—	*	—	*	*	*	1	1	—
<i>Olinga feredayi</i> (McLachlan)	—	—	—	—	—	*	*	*	—
<i>Orthopsycha fimbriata</i> (McLachlan)	—	1	1	1	8	4	*	*	—
<i>Oxyethira albiceps</i> McLachlan	6	1	4	—	—	—	*	*	*
<i>Plectrocnemia maclachlani</i> Mosely	—	—	—	*	—	—	—	6	7
<i>Polyplectropus</i> sp.	—	—	—	—	—	—	*	—	—
<i>Pycnocentrella eruensis</i> Mosely	—	—	*	—	—	—	*	—	—
<i>Pycnocentrodes</i> spp.	—	—	—	—	—	*	*	—	—
<i>Psilochorema</i> spp.	*	1	1	1	2	*	*	1	*
<i>Triplectides obsoleta</i> (McLachlan)	—	—	—	—	—	—	*	1	2
Diptera									
<i>Aphrophila noezelandica</i> (Edwards)	—	—	*	*	—	—	3	—	*
<i>Austrosimulium</i> sp.	*	14	4	5	2	—	*	1	1
<i>Chironomus</i> sp. A	—	—	—	—	—	—	*	—	11
<i>Cricotopus</i> sp.	37	20	37	1	—	1	19	1	*
Diamesinae unident.	—	—	—	—	1	—	—	—	—
Empididae	*	*	—	—	—	—	*	—	—
Ephydriidae	*	—	*	—	—	—	—	—	—
Eriopterini sp.	—	*	—	3	*	3	1	4	3
<i>Eukiefferiella</i> sp.	15	2	17	2	—	—	6	—	*
Macropelopiini	1	2	2	*	*	5	4	12	7
<i>Maoridiamesa</i> sp.	3	—	*	—	—	—	*	—	—
<i>Naonella forsythi</i> (Boothroyd)	14	8	7	1	10	*	6	—	*
<i>Nothodixa</i> sp.	*	3	1	1	*	—	*	1	*
Orthoclad sp.	13	1	*	—	2	—	*	—	—
<i>Paralimnophila skusei</i> Hutton	—	*	*	*	*	—	*	*	*
<i>Paucispinigera</i> sp.	—	—	—	1	—	—	—	—	*
Psychodidae	—	—	—	*	—	—	—	—	—
<i>Polypedilum</i> sp.	1	12	7	1	8	2	3	*	5
Stratiomyidae	—	—	—	*	—	—	—	—	—
Tanyderidae	—	—	—	—	—	1	*	—	—
<i>Tanytarsus vespertinus</i> Hutton	—	*	*	2	*	2	*	1	*
Coleoptera									
Elmidae (larvae)	*	1	*	2	2	3	3	8	8
Elmidae (adult)	—	1	*	—	*	2	*	—	—
Ptilodactylidae (larvae)	—	—	—	*	—	*	—	—	—
Scirtidae	—	1	—	—	—	*	—	2	2
Mollusca									
<i>Latia</i> sp.	—	—	—	—	—	—	*	—	*
<i>Potamopyrgus antipodarum</i> (Grey)	*	1	2	1	1	*	8	4	2
<i>Physa</i> sp.	—	—	—	—	—	—	*	—	—
Oligochaeta									
<i>Eiseniella</i> sp.	*	1	3	3	*	—	1	1	—
Naididae	*	1	—	2	—	—	*	1	*
Nematoda	*	—	—	—	—	—	—	—	—
Crustacea	—	—	—	—	—	—	—	—	—
Ostracoda	*	*	—	*	—	—	19	18	1
<i>Paracalliope fluviatilis</i> (Thompson)	—	—	—	*	—	—	—	—	—
<i>Paratya curvirostris</i> (Heller)	—	—	—	*	—	—	—	—	*
<i>Paranephrops zelandicus</i> (White)	—	—	—	*	—	—	*	—	—
Acarina	*	—	—	*	—	*	—	—	—
Total invertebrates (numbers m ⁻²)	9963	1328	3947	1110	1145	1280	8817	1428	1588
SD	6073	1118	1828	547	121	806	4888	961	1769