Pine afforestation and stream health: a comparison of land-use in two soft rock catchments, East Cape, New Zealand

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(Received 20 January, revised and accepted 10 February 2007)

Abstract

Environmental conditions in streams draining pastoral catchments are typically degraded by comparison with native forest streams. However, few studies have investigated land-use influences on streams in erosion-prone terrain where channel disturbance and mobilisation of land contaminants under pasture might be particularly severe. We studied the water quality and stream ecological 'health' of streams in two catchments in the Gisborne District, eastern North Island, New Zealand, where pine afforestation is proceeding rapidly in an effort to reduce erosion. We compared pasture, pine plantation, and native forest land-uses in these catchments that were characterised by differing forms of erosion of the soft rock hill country. Streams in mature pine plantations in both catchments had generally better water quality (lower faecal contamination and nutrient concentrations) than those in pasture, and tended to approach the condition of reference streams in native forest. However, visual clarity and turbidity and particulate forms of nutrients remained degraded in pine plantations where deeply incised gullies continued to yield large amounts of fine sediment. Stream stability had the dominant influence on epilithon biomass and invertebrate density, and increasing taxa richness was related to increasing stability. Invertebrate community metrics of stream health (%EPT and QMCI) were degraded in pasture compared to pine and native forest, and community composition was influenced by both stability and water temperature. In these streams, establishment of mature pine plantations on pasture in soft rock terrain resulted in water quality and stream health conditions similar to native forest streams.

Keywords: pasture - pine plantation - native forest - water quality - stream invertebrates - turbidity - stability - faecal indicator bacteria - nutrients.

Introduction

Numerous studies in New Zealand have reported that environmental conditions (water quality and ecological 'health') of streams draining pasture are degraded by comparison with those in native forest, and have been reviewed by Smith *et al.* (1993) and Parkyn & Wilcock (2004). However, most studies have been in tectonically stable areas and competent rock types, with consequently moderate erosion rates. Streams in erosion-prone soft-rock areas of New Zealand such as the Gisborne District (Hicks *et al.* 2000; Mazengarb & Speden 2000) may be expected to be severely degraded under pasture land-use, because of their high rates of sedimentation, unstable channels, and often very turbid waters. Comparatively little research has been done on the environmental conditions of such streams compared with streams in areas of the country with more competent rock types (exceptions are Davies-Colley & Stroud 1995; Horrox 1998; and an assessment of fish communities in the East Cape region by Richardson & Jowett 2002). It is not known, for example, to what extent the degraded condition of streams in soft rock country merely reflects the unstable underlying geology, irrespective of land-use, as opposed to the interaction of livestock grazing with susceptibility to erosion.

Deforestation and conversion to pasture are widely recognised to have accelerated erosion of soft-rock hill country (DeRose *et al.* 1993; Page *et al.* 1994) and replanting of degraded pasture land with pines is proceeding rapidly throughout the Gisborne District in an effort to reduce erosion and sediment yields. We studied streams in two catchments affected by different forms of erosion. Erosion in the Te Arai catchment, south-west of Gisborne, was characterised by shallow landslides initiated by storms, whereas the Mangaoporo Valley, near Ruatoria, was prone to gully erosion of fine-grained sedimentary rocks. We were interested in the extent to which pine afforestation of grazed pasture would improve stream condition towards that of (comparatively undisturbed) reference streams in native forest.

In these erosion prone catchments, we compared pine plantation streams with pasture streams (typical land-use prior to planting with pines) and native forest streams (original land cover). We assessed water quality and stream health, using invertebrate communities as indicators, and measured both substratum stability (Quinn & Hickey 1990; Townsend *et al.* 1997b; Death 2002) and shade (Quinn *et al.* 1997; Rutherford *et al.* 1999), which often emerge as major determinants of invertebrate community diversity and abundance. New Zealand native forest streams are frequently heavily shaded and characterised by dense riparian vegetation. Despite stream channels in soft-rock areas being generally unstable, we expected that the ecological health and water quality of streams previously in pasture would be improved by pine plantations, which should restore the shade and temperature functions of native forest, and reduce livestock access to streams.

Methods

Study Sites

We chose stream sites in two major river catchments with some of the very few remaining native forest remnants in the East Cape region, New Zealand (Figure 1). General features of the pasture, pine, and native forest sites are given in Table

1. The sites are ordered according to land-use within each cluster. Pasture catchments were characterised by extensive beef and sheep grazing and the pine plantation catchments were established in 1978; thus, trees were 22 years old at the time of sampling.

One cluster of four stream sites was located in the Te Arai subcatchment of the Waipaoa River catchment, south of Gisborne (Figure 1B, Table 1). Hill slopes were predominantly 26° - 35°, underlying lithologies were inter-bedded fine-grained sandstone and mudstone (Mazengarb & Speden 2000), and erosion was characterised by slips of shallow depth (<1 m), often triggered during periods of intense rainfall such as cyclonic storm events (Marden & Rowan 1993). The native forest headwaters of the Te Arai stream, a water supply catchment for Gisborne, provided a reference site. Two pasture tributaries were sampled, the Waingake Stream and an un-named right bank tributary of the Te Arai River upstream of the Waingake confluence. A tributary of the Te Puninga Stream ('Rimu Stream') in a catchment adjoining the Te Arai catchment, but in the Nuhaka River Catchment (i.e., outside the Waipaoa catchment), provided a catchment in a pine (*Pinus radiata*) plantation (Wharerata Forest).

A second cluster of six stream sites was situated in the Mangaoporo Valley near Ruatoria (Figure 1C; Table 1). The underlying lithologies, consequent landforms and erosion processes, contrasted markedly with those in the southern cluster of stream catchments. In the three Weraamaia catchments (M5, M6 and M10) bedrock consisted of a basal noncalcareous mudstone overlain by well-bedded calcareous mudstones. Hill slopes were steeper than in the southern catchments and were susceptible to extreme gully erosion. The remaining catchments (M7, M8 and M9) had lithologies of alternating sandstone and mudstone that were typically more highly tectonised (crushed). Consequently, the predominant erosion processes in catchments M7, M8 and M9 were moderate to very severe earthflows and severe to very severe gully erosion (Mazengarb & Speden 2000). One native forest catchment provided a reference stream, two catchments were in pastoral land-use, two were in mature pine plantations, and one had mixed land-use (mainly *P. radiata* but some native forest) (Table 1). One of the two Mangaoporo pasture sites (Mangatekapua Stream, M8) was closed to livestock and planted in *P. radiata* during the one-year study (commencing July 1999, 2 months after sampling began).

Water quality samples were taken monthly for 14 months (12 months at two Mangaoporo Valley sites). Macroinvertebrate communities and epilithon were sampled quarterly: winter, 26-28 July 1999; spring, 26-27 October 1999; summer, 1-3 February 2000; and autumn, 29-31 May 2000. On one visit (February) a comprehensive survey of stream habitat characteristics was conducted.

Hydrology

In the Te Arai catchment an estimate of flow at each site (or at a nearby site used as a flow reference) was obtained by current meter gauging (Small Ott, Ott Co., Germany) on each monthly visit. The estimated instantaneous flows on sampling visits to the Te Arai sites were correlated with simultaneous flows at a continuous water level recorder at Pykes Weir on the Te Arai River (Figure 1B).

In the Mangaoporo Valley, current meter gaugings were made monthly with the same instrument at two road bridges on Mangaoporo Valley Road. The Mangatekapua Stream at a road bridge immediately downstream of the point where the right branch (M7) and left branch (M8) join (Figure 1C), provided a reference for flow in these two true-right tributaries and also for the Mangaehu Stream (M9). The Weraamaia Stream at a bridge on Mangaoporo Valley Road on the true-left side of the Mangaoporo River (Figure 1C) provided a reference for flow in the headwater tributaries M5, M6 and M10. No continuous flow recorder stations are operated in the immediate vicinity of the Mangaoporo River, however, the hydrograph for the Waiapu River at Ruatoria, just upstream of the Mangaoporo confluence, provided a general indication of state of flow on sampling visits.

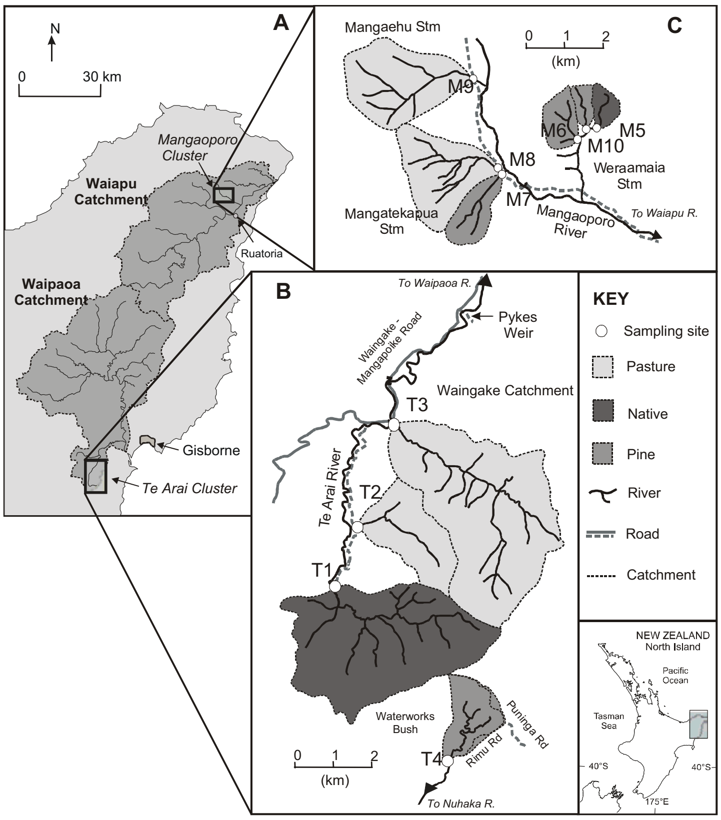

Figure 1. A, location map showing the two clusters of study sites in the Gisborne District, eastern North Island; B, catchments of study streams in the Te Arai Catchment; C, catchments of study stream sites in the Mangaoporo Catchment.

Table 1. Locations of the 10 study sites and characteristics of their catchments. Flow values in parentheses were estimated from flows at the Mangatekapua or Weraamaia Bridges on Mangaoporo Valley road (see text).

Stream Site	Site code	Map reference (NZMS260)	Altitude (m)	Catchment area (km²)	Median flow (l s^{-1})	Land-use
Mangaoporo Valley (near Ruatoria)						
Mangaehu Stream	M9	Z15 715 642	160	4.6	(25)	Pasture
Mangatekapua Past Stream	M8	Z15 718 619	140	3.5	(23)	Pasture, planted in pines late 1999
Mangatekapua Pine Stream	M7	Z15 718 619	140	1.5	(10)	Mature *P. radiata*
Weraamaia Pine Stream	M6	Z15 743 625	160	1.0	(20)	Mature *P. radiata*
Weraamaia Mix Stream	M10	Z15 745 628	200	0.6	(10)	Mature *P.radiata*, some native forest
Weraamaia Native Stream	M5	Z15 749 628	240	0.7	(14)	Native forest (reference)
Te Arai Catchment (SW of Gisborne)						
Waingake Stream	T3	X18 270 573	60	12.8	140	Pasture
Un-named tributary of Te Arai	T2	X18 262 547	140	3.8	43	Pasture
'Rimu' Stream	T4	X19 288 491	420	1.9	37	Mature *P. radiata*
Te Arai River	T1	X18 257 525	200	10.4	220	Native forest (reference)

Water Quality

Conductivity was measured on-site (YSI model 30 meter) until the meter was stolen in March 2000 after eleven monthly visits. Visual clarity of stream water was measured as the sighting range of a black disc observed horizontally underwater (Davies-Colley 1988). No direct reading could be obtained on some occasions of high visual clarity in very small streams, when the sighting distance was greater than the range available in pools within the channel.

Water samples were taken from each site on each visit in 110 ml sterile vials for enumeration of the favoured faecal indicator bacterium *E. coli* by the Colilert test kit (IDEXX Laboratories, USA). The method detects *E. coli* biochemically as a subset of total coliforms (Covert *et al.* 1992) and has been shown to agree well with more traditional methods in several studies (e.g., Eckner 1998). Because this 'most probable number' (MPN) method has high precision (with 48 small and 49 large wells on each plate), we report *E. coli* concentrations as cfu/ 100 ml. Samples were transported in insulated bins (dark, chilled) to the NIWA laboratory, and were analysed within about 50 hours of collection.

A pair of 1 l water samples was also taken from each site on each visit in acid-washed polyethlyene bottles for other water quality analyses following APHA (1998). These water samples were handled the same way as the bacterial samples, and analyses commenced on the day of receipt at the laboratory or shortly after for: electrical conductivity, turbidity (Hach 2100AN nephelometer), suspended sediment, and dissolved humic matter expressed as g_{440} m^{-1} values (Davies-Colley & Vant 1987; Kirk 1976; Smith *et al.* 1997). The following nutrient analyses were conducted using the QuikChem FIA+ 8000 series, Lachat Instruments, nutrient analyser (following methods of APHA 1998): dissolved reactive phosphorus (DRP: precision at 5-10 ppb level, ±1 ppb), total phosphorus (TP: at 10-20 ppb, ± 1 ppb; at 500 ppb, ± 10 ppb), ammoniacal nitrogen (Am-N: at 2-10 ppb, ± 1 ppb), nitrate plus nitrite nitrogen (NO_x-N: at 10-20 ppb, ± 1 ppb; at 150 ppb, ± 2 ppb), and total Kjeldahl nitrogen (TKN: at 10-20 ppb, ± 1 ppb; at 440 ppb, ± 40 ppb).

Stream habitat and biotic sampling

Channel stability at each site was evaluated visually by a single observer (MRS) 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. Temperature loggers, recording every 15 minutes, were deployed in pools in February 2000 (late summer) and left for 22 days to record stream thermal regimes during summer.

Particle sizes of streambed substrata (including woody debris) in the wetted channel were sampled using the pebble count method (Wolman 1954) on 10 evenly spaced transects within a 100 m reach. Suspendable inorganic sediments (SIS) and benthic particulate organic matter (BPOM) in the surface sediments were sampled on seven randomly selected transects in each study reach. Bed sediments were stirred with a pole in an enclosed cylinder (diameter 24 cm) to a depth of about 5 cm so as to suspend interstitial fines in the overlying volume of stream water. When the suspendable fines were fully mixed with the water, a *c.*100 ml sub-sample was taken for dry mass (DM, 105 °C) and ash-free dry mass (AFDM, 400 °C for a minimum of 6 h) analysis. SIS and BPOM (g m^{-2} of streambed surface) were calculated from the cylinder area and the enclosed water volume (determined from water depth measurements at 10 points within the cylinder).

Exposure of the study reaches to light was characterized using a canopy analyser (LAI-2000, Li-Cor Inc, Lincoln, Nebraska). The instrument design, based on fish-eye lens imaging, is described by Welles & Norman (1991). We used the protocol described by Davies-Colley & Payne (1998), which involved taking a point reading of the distribution of shade at each of 20 points along the 100 m stream reach (5 m intervals), at both (low) water and bank level. A reach-averaged index of lighting, the diffuse non-interceptance (DIFN), which is close to actual lighting under overcast conditions, was calculated from the point measurements.

In each quarterly survey from July 1999 to May 2000, epilithon and benthic invertebrate samples were taken. Epilithon total biomass (AFDM) and chlorophyll-*a* (90% acetone extraction followed by spectrophotometric measurement with phaeophytin correction; APHA 1998) were measured from five medium-large gravels ('b' axis 16–32 mm) and five small cobbles ('b' axis 64–128 mm) that were taken at random from five of the 10 transects located along the 100 m reach. Epilithon was removed with a nylon-bristled brush into a known volume of water. Epilithon biomass and chlorophyll-*a* content were expressed per unit area of exposed stone surface (assumed to be half the total surface area; Dall 1979) that we calculated by measuring stone dimensions. Epilithon was measured on both small stones and large stones to determine whether presumed differences in the stability of particles differing in size affected epilithon biomass. Abundance of epilithon was also assessed visually (none, slippery, obvious, abundant, excessive; e.g., Jowett & Richardson 1990) on monthly visits to the streams.

Invertebrates were collected in ten 0.1 m^2 Surber samples (250 µm mesh) taken at 10 m intervals within the 100 m study reach at each site in each season. Samples were preserved immediately in 70% isopropyl alcohol and combined. Composite samples were sorted by eye in a white tray and invertebrates were identified to species level (or morphospecies) where possible using the keys of Winterbourn (1973) and Winterbourn & Gregson (1989). Oligochaeta were identified to family level (Brinkhurst 1971) and Chironomidae to genus or species level where possible (I.K.G. Boothroyd, unpublished key).

Analyses

Probability plots of all data were inspected before analysis and data were log or arcsine square root transformed as appropriate to homogenise variances. For SIS and BPOM data that were taken as random replicated samples on one occasion, we used a nested ANOVA design where sites were nested by geographic cluster to test the differences between both clusters and sites for each variable tested. Tukey's *post-hoc* tests were used to determine where differences lay following a significant ANOVA. For invertebrate metrics where composite samples were taken over time, we used a repeated measures ANOVA in the statistical package NCSS 2000 (Hintze 1998) to test for differences in land-use and cluster. Significance was accepted at $P<0.05$ in all tests. Bonferroni Multiple Comparison tests were used to identify differences among groups.

The Quantitative Macroinvertebrate Community Index (QMCI) was calculated for the invertebrate data using the scoring system of Stark (1993). The invertebrate data set was collapsed to the appropriate taxonomic resolution. Spearman rank correlation ($r_s > 0.65$, $P < 0.05$, $n = 10$) was used to investigate correlations between invertebrate indices and many of the habitat and water quality variables measured for the 10 streams. Canonical Correspondence Analysis (CCA; PC-ORD 3.0) was used to discriminate sites based on invertebrate community structure (February samples only) and to correlate it with the following environmental variables: daily maximum temperature, epilithon biomass (chlorophyll-*a*), turbidity, and stability (Pfankuch index). We chose variables that were considered to have high relevance for invertebrate communities.

Results

Water Quality

All sites were on small streams having median flows ranging from 10-220 l s^{-1} (Table 1). Median water yields were 7-20 l s^{-1} km^{-2}.

Conductivity (Table 2; Figure 2A) varied appreciably between the two clusters of streams with sites apparently reflecting differences in land-use. Median conductivity values were comparatively low at the native forest sites in both clusters (medians of 140 and 300 µS cm^{-1} at 25 °C), however values were more variable at pine plantation sites. They were comparatively low at Te Arai cluster (median 190 µS cm^{-1}) but high at one of the pine sites in the Mangaoporo cluster (M7, median 600 µS cm^{-1}) where there was disturbance and erosion in the catchment. Conductivity at pasture sites (390-460 µS cm^{-1}) was similar to that at pine sites in the Mangaoporo cluster, but it was two-fold higher in pasture streams than pine plantation streams at Te Arai.

Suspended sediment concentration (SSC) and related variables (visual clarity and turbidity) varied markedly among sites, and were apparently more strongly associated with geological differences than land-use (Table 2). The box plot for turbidity (Figure 2B) had a very similar pattern to that for SSC (not shown). In the Te Arai catchment, the two pasture sites were more turbid than the pine plantation or native forest sites, but were less turbid than the pasture streams in the Mangaoporo catchment (M8 and M9). These latter streams had median black disc visibilities of only a few centimetres (Table 2) and turbidities in the 100s of NTU. One pine plantation site (M7), another right-bank tributary of the Mangaoporo River, was also very turbid (medians of 7 cm visibility and 100 NTU), apparently due to active gullying in its catchment. Streams in the Weraamaia Catchment on the true left bank of the Mangaoporo River (pine and native forest catchments) were much less turbid (generally <10 NTU), and their water was sometimes too clear for measurement using the black disc technique.

The aquatic humus ('yellow substance') content of the streams was moderate to high on a New Zealand wide basis (Davies-Colley & Close 1990; values of $g_{440} > 1$ m^{-1} can be regarded as "high" and values <0.1 m^{-1} as "low") and showed a weak tendency to be higher in pasture than in forest catchments (Table 2).

Table 2. Median values for 14 water quality variables measured at 10 stream sites in the Mangaoporo and Te Arai catchments in 1999-2000. Also shown are median values for the NZ Rivers Water Quality Network (NRWQN) (Smith & Masdaam 1994; Smith *et al*. 1997). SS = suspended sediments; forms of nitrogen as defined in the text.

Site code	Conductivity (μS cm^{-1})	Clarity (m)	Turbidity (NTU)	SS (g m^{-3})	Humic substance g_{440} (m^{-1})	*E. coli* (cfu/100 ml)	Nitrogen (mg m^{-3}) Am-N	NOx-N	DIN	TKN	OrgN	TN	Phosphorus (mg m^{-3}) DRP	TP
Mangaoporo														
M9 pasture	415	0.095	90	110	0.53	52	12	71	94	200	200	250	3.3	96
M8 pasture	458	0.034	580	700	0.98	48	59	140	190	450	510	610	3.0	300
M7 pine	596	0.073	100	100	0.78	120	22	120	140	250	230	350	3.1	130
M6 pine	394	3.5	0.80	1.2	0.30	8	3.0	10	14	38	52	66	4.0	7.0
M10 mix	369	0.27	4.6	16	0.34	17	2.0	11	14	54	70	97	3.0	13
M5 native	141	5.0	1.05	1.7	0.44	30	3.0	15	15	50	57	73	5.8	9.0
Te Arai														
T3 pasture	393	1.05	5.3	5.6	0.77	180	4.0	21	33	150	160	210	2.4	11
T2 pasture	453	0.40	14	19	0.69	140	3.0	19	28	160	160	180	2.2	16
T4 pine	185	4.2	0.95	1.5	0.55	120	3.5	160	160	68	71	200	4.6	9.0
T1 native	295	2.5	2.2	2.1	0.59	45	4.0	17	23	64	74	96	4.7	13
NRWQN	85	1.3	2.4		0.72		9	105	114			245	4.0	17

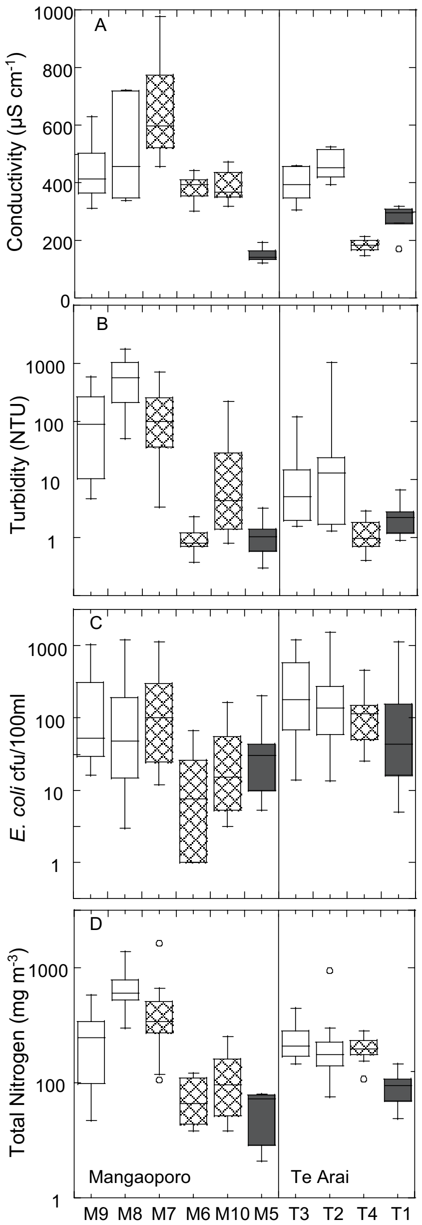

Figure 2. Box plots of A, conductivity; B, turbidity; C, *E. coli* and D, total nitrogen in the 10 study streams. Closed boxes are native forest streams, cross-hatched boxes are pine plantation streams, and open boxes are pasture streams. Note that data for turbidity, *E. coli* and total nitrogen are on logarithmic axes. Boxes show the median and interquartile ranges of the data; lines extend to the maximum or minimum points that fall within the boundary of 1.5 x the interquartile range; circles represent outliers.

Faecal contamination, as indicated by *E. coli*, showed a comparatively weak association with land-use (Figure 2C). In the Te Arai area, median *E. coli* values exceeded 100 cfu / 100 ml in the pasture streams and the pine plantation stream, but the median value was lower in the native forest stream (although not particularly low in absolute terms, median 45 *E. coli* cfu / 100 ml for site T1). Streams in the Weraamaia Catchment on the true left bank of the Mangaoporo River (pine plantation sites M6, M10, and native forest site M5) had lower *E. coli* concentrations than those of the right bank tributaries and the Te Arai streams. The appreciable amount of faecal contamination in native forest streams in both the Te Arai and Mangaoporo areas, and the high faecal contamination of the pine plantation stream (T4), can probably be attributed to the presence of feral goats and pigs (observed on some visits), and of cattle that had access to the pine block (M7) in the Mangaoporo area. While faecal contamination of pasture sites in the Te Arai area was generally higher than in forest streams over the 14 months that data were collected, the median value was only a little higher than the guideline value for swimming (126 *E. coli* / 100 ml; Smith *et al.* 1993).

Nutrients

A broadly similar pattern was evident across the sites for all forms of nitrogen (ammoniacal [Am-N], nitrate + nitrite [NOx-N], dissolved inorganic [DIN], total Kjeldahl [TKN], organic [org-N] and total [TN]; Table 2). Figure 2D shows box plots for total nitrogen (TN) by way of example. Highest concentrations occurred in the right bank tributaries of the Mangaoporo River (M7, M8, M9). Lowest nitrogen concentrations were found in the forested Weraamaia catchment streams (M5, M6 and M10) and in the native forest stream in the Te Arai area.

Total phosphorus (TP) concentration was highly variable at the Mangaoporo sites (Table 2), and showed a pattern of high concentration where suspended sediment was high (r_s = 0.96). Dissolved reactive phosphorus (DRP) made up a small proportion of total P and did not show the clear difference between the two sides of the Mangaoporo Valley that was evident for the other nutrients. Overall, there was little variation in median DRP concentration among sites (3.6 ± 1.7 mg m^{-3}).

Stream habitat and biota

Stability and temperature

The Mangaoporo sites were less stable than those in the Te Arai cluster (Table 3). In both areas, the most stable streams were in native forest catchments and the least stable were in pasture catchments; pine plantation sites generally had intermediate stability. Some of the Mangaoporo sites had extremely wide channels (notably pasture site M9, *c.* 60 m, Table 3) similar to those of braided, glacial out-wash streams. Moderate channel widths (6-9 m) were also associated with relatively high light exposure (%DIFN; Table 3) at some sites within forest (e.g., M6 [pine] and T1 [native]).

Median stream water temperatures were similar (15-18 °C) across all land-uses. In the Te Arai cluster, pasture sites were warmest and had the greatest range of temperature extremes. Median and maximum temperatures were cooler in the small pine plantation stream than in the larger native forest stream, reflecting differences in channel width, and consequently, light exposure. Maximum temperature during February was very high at the pasture sites in both catchments (>30 °C in M9 and M8; Figure 3). In the Mangaoporo catchment, one of the pine plantation sites also experienced very high (>25 °C in M7 and M10) daily maxima on occasions. The

Table 3. Values for stream habitat variables measured at 10 sites in February 2000. SIS = suspendable inorganic solids, BPOM = benthic particulate organic matter. Superscript letters show differences among sites determined by Tukey's tests following ANOVA. Values with the same letter are not significantly different from one another within columns.

Site code	Pfankuch stability score	Channel width (m)	Light exposure (%DIFN)	% silt / sand (< 2mm)	% gravel (16-32 mm)	% cobble (64-128 mm)	SIS (g m^{-2})	BPOM (mg m^{-2})
Mangaoporo								
M9 Pasture	111	62.8	84.4	3.5	16.8	13.3	937^{abc}	34.4^{abc}
M8 Pasture	111	11.1	30.8	43.4	9.4	6.6	2470^{a}	58.5^{ab}
M7 Pine	98	9.2	15.2	21.0	19.0	6.7	1620^{ab}	40.5^{ab}
M6 Pine	92	6.4	59.5	6.3	20.5	10.7	390^{cd}	20.4^{bc}
M10 Mix	88	4.8	35.9	7.4	13.9	7.4	586^{bc}	18.2^{bc}
M5 Native	64	5.5	14.4	15.9	13.3	11.5	242^{d}	14.1^{c}
Te Arai								
T3 Pasture	77	7.8	41.4	12.5	3.8	27.9	2050^{a}	86.9^{a}
T2 Pasture	74	4.8	63.5	2.9	7.8	32.0	1860^{a}	69.8^{a}
T4 Pine	69	3.6	2.4	16.9	5.9	14.4	1040^{abc}	78.6^{a}
T1 Native	57	8.2	53.7	9.8	4.9	15.7	1890^{a}	75.8^{a}

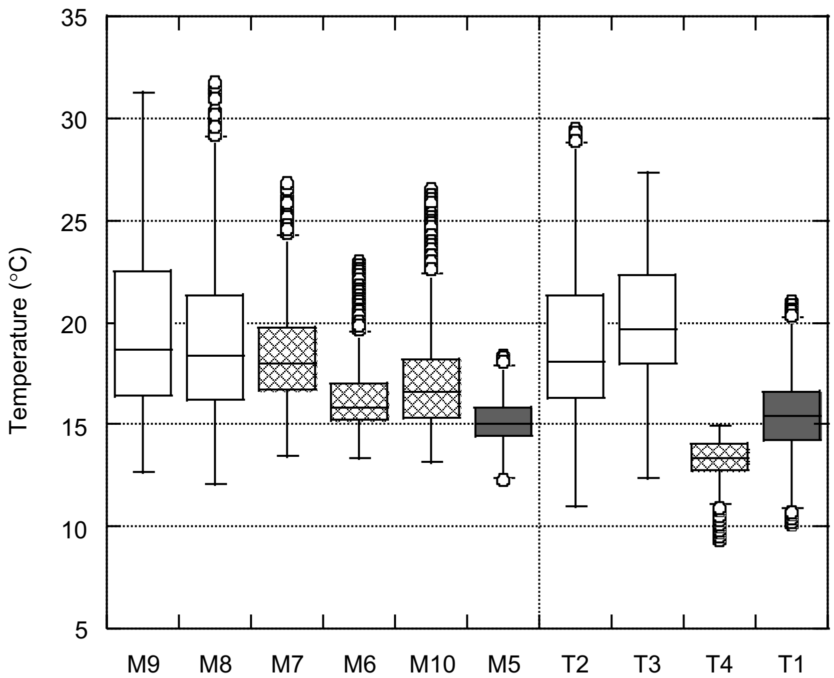

Figure 3. Box plots of temperature measured with data loggers for 22 days in February 2000. Closed boxes are native forest streams, cross-hatched boxes are pine plantation streams, and open boxes are pasture streams. Plots as in Figure 2.

lowest maximum temperature was recorded at the native forest site. Although sites M7 (pine) and M5 (native) had comparable channel widths and light exposure, the stream channel throughout the M7 catchment was affected by gully erosion initiated under pasture, and this may have increased overall light exposure and water temperature.

Substrata and suspended matter

Substratum size distribution differed between the two clusters, the much greater proportion of cobbles and boulders in the Te Arai streams than in the Mangaoporo streams reflecting geological differences (Table 3). The parent rock of the Mangaoporo catchments was weaker and more susceptible to physical weathering than the rounded sandstone cobbles and boulders at the Te Arai sites. There was little difference in the sizes of streambed substrata in catchments with different land-uses, particularly in the Te Arai cluster, although the pasture site M8 had the highest proportion of fine sediment (43% sand and silt; Table 3). The high proportion of fines at some Mangaoporo sites may have been due to the availability of readily transportable, fine-grained bedload in the riverbeds further upstream. Woody debris contributed little to streambed habitat, except at pine site T4.

Benthic particulate organic matter (BPOM) and suspendable inorganic sediments (SIS) were found in significantly greater amounts at the Te Arai sites than the Mangaoporo sites (BPOM $F_{(1,49)} = 57.3$, $P < 0.001$; SIS $F_{(1,49)} = 24.9$, $P < 0.001$; Table 3). Differences among individual sites within catchments were also highly significant (BPOM $F_{(8,49)}$ = 57.3, P = 0.005; SIS $F_{(8,49)}$ = 7.2, P <0.001). BPOM and SIS concentrations were highly variable at the Mangaoporo sites, with highest values at M8 (pasture) and M7 (pine) and lowest values at M5 (native). In contrast, concentrations were consistently high and not significantly different among the Te Arai sites (Table 3).

Epilithon

Epilithon was seldom visually abundant on monthly visits to the Mangaoporo streams, except at pine site M6, however in the Te Arai cluster it was variably abundant and sometimes "excessive" in the two pasture streams. Measured epilithon densities were generally lowest in July 1999 (winter) and highest in summer, but as seasonal patterns were unclear mean chlorophyll-*a* data are presented in Figure 4. Native forest stream (T1) was comparatively large with a distinct canopy gap (54% DIFN, Table 3), and epilithon abundance was greater there than in the heavily-shaded, small pine plantation stream (T4).

Mean chlorophyll-*a* concentration was much greater at the Te Arai than Mangaoporo sites ($F_{(1,38)}$ = 37.1, P = 0.004; Fig. 4), and the low levels at all Mangaoporo sites were consistent with their channel (and substratum) instability as indicated by the Pfankuch index. No significant difference in chlorophyll-*a* concentration was found among land uses ($F_{(2,4)}$ = 0.01, P = 0.989). The amount of chlorophyll-*a* was similar on gravels and cobbles across sites suggesting similar amounts of physical abrasion regardless of substratum size. The Te Arai native forest site (T1) had moderately high light levels (%DIFN; Table 3) that may have promoted epilithon growth, as the study reach was relatively open from a lightly vegetated floodplain on one side. However, the Te Arai pine site (T4) had very high levels of chlorophyll-*a* considering it was heavily shaded. Field observations suggested that some moss might have been included in the T4 epilithon samples, and if so would have contributed to the high chlorophyll-*a* levels. The Mangaoporo pine plantation stream (M6) had relatively high chlorophyll-*a* concentrations that may have been related to the higher than average clarity of the water (Table 2).

Invertebrates

Invertebrate density ($F_{(1,4)}$ = 23.2, P = 0.008), taxa richness ($F_{(1,4)}$ = 37.8, P = 0.003), and QMCI ($F_{(1,4)}$ = 7.7, P = 0.049) values were significantly greater at the Te Arai sites than the Mangaoporo sites (Figure 5). Land-use did not affect invertebrate density ($F_{(2,39)}$ = 1.6, P = 0.295) (Fig 5a), although the highest invertebrate density was recorded at pasture site T3 in July 1999 (mean 23480 m^{-2}) when midge larvae (Chironomidae) predominated (96% of the sample). In contrast, invertebrate densities at the two Mangaoporo pasture sites were very low (116 and 138 m^{-2}, respectively). Significant differences were found between land-uses for % EPT taxa ($F_{(2,4)}$ = 25.8, P = 0.005) and QMCI ($F_{(2,4)}$ = 44.7, P = 0.001) (Figure 5c,d). These metrics were higher at the forest sites than the pasture sites, and those for the pine forest sites did not differ significantly (P >0.05) from those for the native forest sites. Mean QMCI values for pine and native forest sites were above the level indicative of "clean water" (Stark 1993), whereas all of the pasture sites were regarded as having "probable severe or

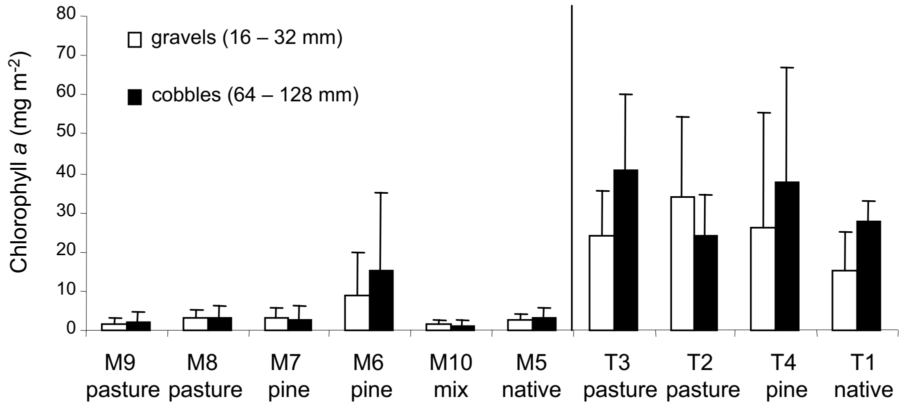

Figure 4. Epilithon measured as chlorophyll-a concentration from gravel and cobble substrata taken from the 10 stream sites. Data shown are means (±1 SD) of four seasonal samplings.

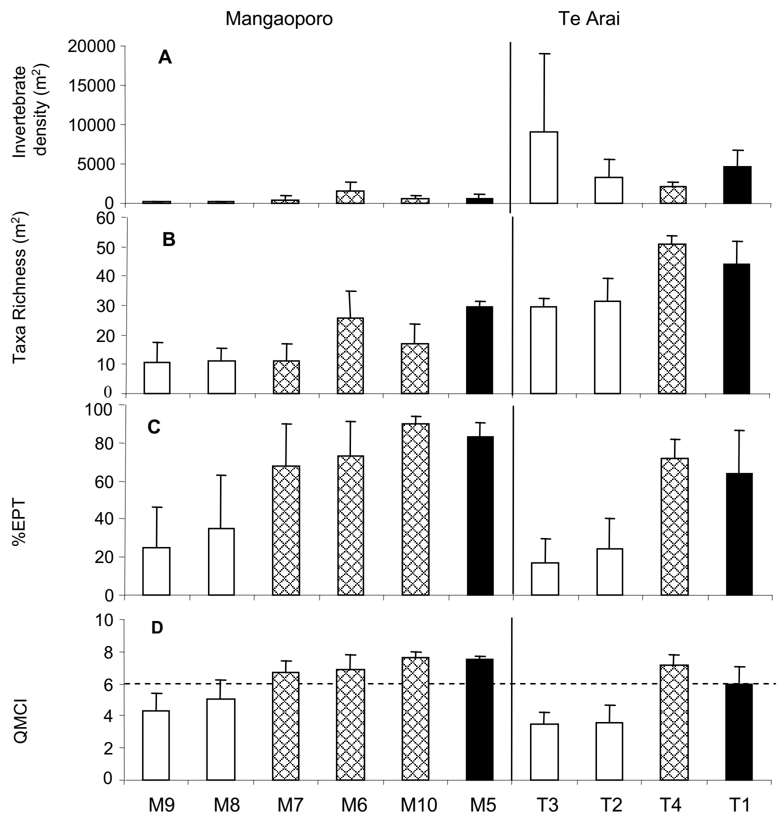

Figure 5. A, macroinvertebrate abundance; B, taxonomic richness; C, percentage of individuals of Ephemeroptera, Plecoptera and Trichoptera (%EPT) and D, the Quantitative Macroinvertebrate Community Index (QMCI). Bars show means (±1 SD) of four seasonal samples. Closed boxes are native forest streams, cross-hatched boxes are pine plantation streams, and open boxes are pasture streams. The dashed line in D is the lower threshold for clean water (Stark 1993).

moderate pollution".

Taxonomic richness was low at the three unstable sites M7-9 (Figure 5b), but while the pattern of differences between land-uses was similar to those for %EPT taxa and QMCI, differences in taxa richness were not significant (P >0.05). Stoneflies, which are sensitive to high water temperature (Quinn *et al.* 1994), were not found at the Mangaoporo pasture sites, and were extremely rare (2 individuals) at the Te Arai pasture sites. Stoneflies were commonly found in native forest, however, with high numbers of *Zelandoperla decorata* (up to 144 m^{-2}) at the Te Arai site. Pine forest streams were also inhabited by stoneflies, although fewer species and at lower densities than in native forest streams. Overall, 10 species of Plecoptera were recorded. Diptera, which are generally tolerant of environmental disturbance, dominated pasture stream communities.

Community composition at pine plantation site T4 differed strongly from all other sites (Figure 6). Species that were present at T4 and not at other sites included several typically associated with small, forest streams (e.g., *Oeconesus maori* and *Triplectides obsoletus*) and larvae associated with aquatic bryophytes (*Zelolessica cheira*). CCA indicated that high epilithon biomass (that might have included moss) may explain these differences.

All other sites were separated in the CCA along a gradient of stability and water temperature (Figure 6). Invertebrate communities of the cool, stable, native forest sites (both clusters) clustered near the top of the diagram. Mangaoporo pine plantation sites clustered together with the Te Arai pasture sites and M8 (pasture), and were intermediate along the stability gradient. The invertebrate community of the Mangaoporo pasture site M9 (the site with the highest daily maximum temperatures

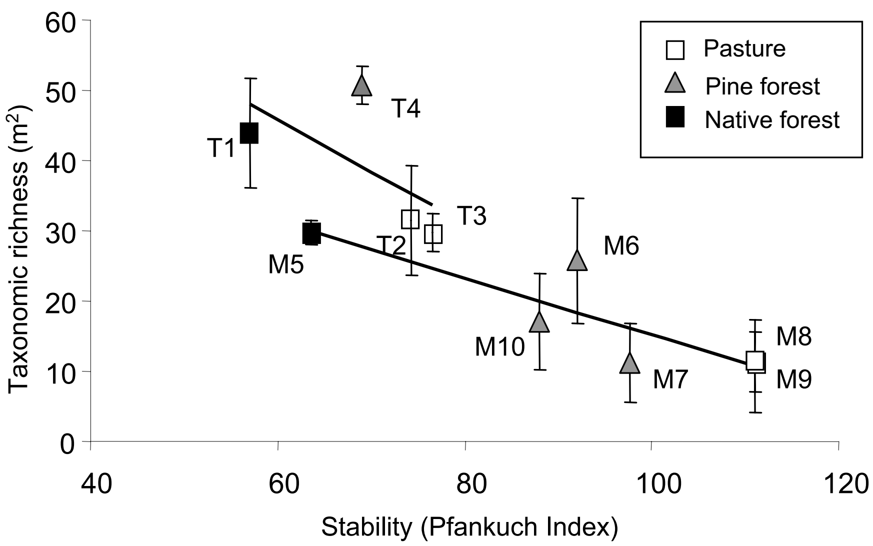

Figure 6. Canonical correspondence analysis (CCA) plot of invertebrate communities at ten sites in February 2000. The lines show the relationship between the sites and three explanatory environmental variables (daily maximum temperature, epilithic algal biomass (chl-a), and stream stability [Pfankuch Index]) in ordination space. The length of line indicates strength of correlation.

and lowest stability), showed the greatest separation from the native forest sites.

Relationships among variables

Taxonomic richness of invertebrates was strongly and inversely correlated with the Pfankuch Index of stream stability in both geographic clusters (Figure 7). The Pfankuch Index was also significantly and negatively correlated with invertebrate density across all sites (r_s = -0.75). Measures of stream health (QMCI) and sensitive taxa (%EPT) were correlated significantly with mean water temperature (-0.78 and -0.75, respectively), but not with stability (-0.23 and -0.18, respectively). Epilithon (chl-*a*) was also correlated significantly with invertebrate density (0.79) and taxa richness (0.83). Suspended solids, turbidity and visual clarity were all strongly intercorrelated ($r_s > \pm 0.9$) and TN, TP, and conductivity were also correlated with turbidity (r_s = 0.85, 0.96 and 0.78, respectively). The only water quality variables correlated significantly with invertebrate metrics were suspended solids (with taxonomic richness; -0.68) and organic nitrogen (with %EPT; -0.65).

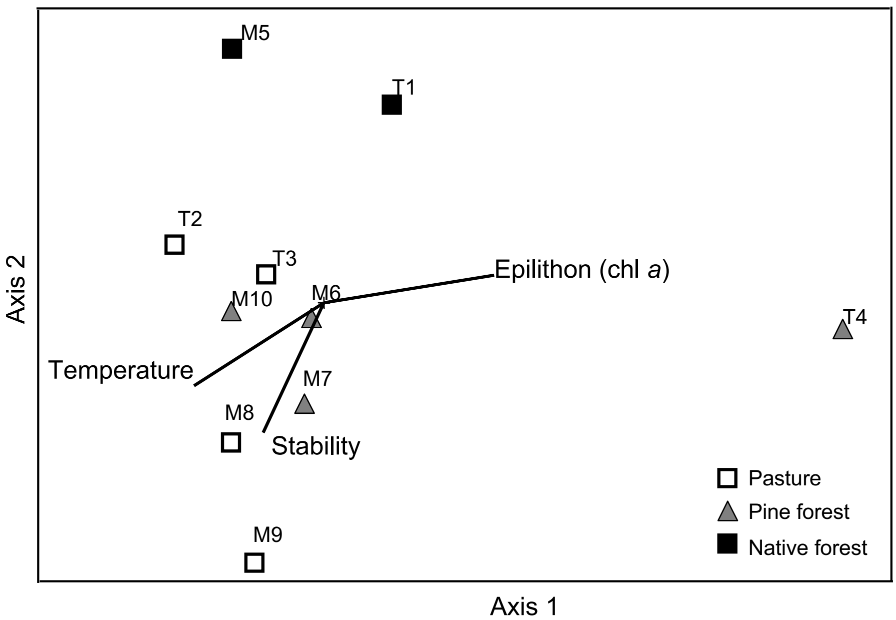

Figure 7. Relationship between invertebrate taxonomic richness and the Pfankuch stablity index for stream sites within the two geographic clusters. Symbols are means (±1 SD) calculated from four seasonal samples. For the Te Arai sites r^2 = 0.41, and for the Mangaoporo sites r^2 = 0.75.

Discussion

Water quality

Streams in the Gisborne District drain catchments of soft rock terrain and differ from many other streams in New Zealand where comparisons of land-use have been made, because of high sediment yields, high water turbidity, and channel instability. Median turbidity ranged from 1 NTU at some pine plantation and native forest sites to 580 NTU in a pastoral stream with a wide, unstable channel (M8). New Zealand rivers generally have much clearer water (median black disc clarity = 1.30 m, median turbidity = 2.4 NTU; Smith *et al.* 1997) and the guideline for swimming water quality is >1.6 m clarity (Ministry for the Environment 1994). In a study of pasture, pine and native forest land-use on Waikato hill country streams, Quinn *et al.* (1997) reported median turbidities of <20 NTU at all sites, although values were highest for pine plantation streams and were attributed to bank erosion.

The Te Arai forested sites, and the Weraamaia catchment sites (M5, M10, M6) had relatively clear waters, but the pasture sites in the Te Arai area were comparatively turbid, and the right-bank tributaries of the Mangaoporo River had extremely turbid waters. The pattern of turbidity appeared to be more strongly related to geological factors than land-use. The high turbidity of streams with active gully erosion initiated after the conversion of large tracts of native forest to pasture in the 1920s (Taylor 1967) may not improve until these gully features are arrested by pine afforestation. Studies by one of us (MM) suggest that this erosional process is scale-dependent: erosion in small gullies (<20 ha) can be arrested within one pine rotation (*c.* 28 y) but in large gullies (>20 ha) it may be very long-lived.

Other water quality variables also appeared to be related to erosion. The comparatively high conductivity compared to freshwaters elsewhere in New Zealand (nationwide median *c.* 85 m K^{-1} at 25 °C; Smith & Maasdam 1994), particularly in the turbid right-bank tributaries of the Mangaoporo River, may reflect high rates of physical erosion that continually expose fresh rock to weathering and leaching (conductivity was correlated with SS, turbidity and visual clarity; r_s = 0.73, 0.78, -0.84, respectively). Total phosphorus was extremely variable among the Gisborne streams but showed a clear pattern of high concentration where suspended sediment was high (r_s = 0.96). The high levels of TP in turbid tributaries were mostly in particulate form, (as for example, in streams of the Whatawhata area, Waikato (Quinn & Stroud 2002). Nitrate nitrogen and ammonical-N were appreciably lower than in New Zealand rivers, generally (nationwide medians = 105 and 9 mg m^{-3}, respectively; Smith & Maasdam 1994), but were elevated in the true right bank tributaries of the Mangaoporo River. Dissolved inorganic nitrogen (nitrate plus ammoniacal-N) and dissolved reactive phosphorus were generally lower than guidelines to control nuisance periphyton growth (100 mg m^{-3} DIN, 15-30 mg m^{-3} DRP; Ministry for the Environment 1992), and consistent with this, epilithon biomass was low to moderate.

Many studies have found that faecal bacterial concentrations are often appreciably higher in pasture than forest catchments and tend to increase with stocking rate or the proportion of farming within the catchment (Smith *et al.* 1993; Davies-Colley & Stroud 1995; Collins 2003). The similarity in faecal contamination status of streams in forest and pasture in the Gisborne District was therefore surprising, and we speculate that it may reflect (a) high populations of feral animals in forests and plantations, (b) relatively low livestock densities in pasture and (c) comparatively rapid bacterial inactivation by sunlight in (unshaded) pasture streams.

Overall, water quality of native forest reference streams (T1, M5) was high, as expected, although perhaps not as high as in hard-rock areas of the country, possibly because of the presence of feral mammals and straying livestock. The water quality of mature, first rotation pine plantations was generally closer to that of native forest streams than pasture streams, suggesting appreciable improvement with reforestation. However, high sediment yields and turbidity persisted where gullying had occurred.

Stream habitat and biota

Studies of land-use change often report higher invertebrate densities in pastoral or open streams compared to forest streams (Harding & Winterbourn 1995; Friberg *et al.* 1997). Quinn *et al.* (1997) found invertebrate densities were 3-fold higher in pasture streams than forested reaches of Waikato hill-country streams, but the mean invertebrate densities at Mangaoporo sites M8 and M9 (116 & 138 m^{-2}, respectively) were very low compared with those reported from equivalent-sized pasture streams elsewhere in New Zealand. Furthermore, in contrast to other studies of land-use effects, we found no significant difference in invertebrate density between forest and pasture streams.

Higher invertebrate densities in pasture than forest streams have sometimes been attributed to their higher primary productivity. Quinn *et al.* (1997), Friberg *et al.* (1997), Townsend *et al.* (1997a), and Harding *et al.* (1999) all found that an increase in invertebrate density accompanied an increase in chlorophyll-*a* concentration in pasture streams. We found no association between epilithic chlorophyll-*a* or invertebrate density and land-use in our study, although invertebrate density was correlated significantly with chl-*a* concentration ($r_s = 0.79$, $P < 0.05$).

Geographic differences in algal epilithon appear to be controlled by substratum instability in our streams. Thus, at the comparatively stable Te Arai sites, mean chlorophyll-*a* concentrations (all > 20 mg m^{-2}) were greater than those in the Mangaoporo cluster (all <15 mg m^{-2}). A number of factors control growth of algae, including light, nutrients, physical abrasion, and invertebrate grazing (Biggs 2000). Light clearly differed between most pasture and forested sites, but the biomass of algal epilithon at the Mangaoporo pasture sites (M8 & 9) was extremely low for nutrient-replete, light-exposed sites. In the Waikato region, pasture streams had much higher chlorophyll-*a* concentrations (16 mg m^{-2}) than native forest and pine plantation streams (<1 mg m^{-2}; Quinn *et al.* 1997), and chlorophyll-*a* was >80 mg m^{-2} in open streams compared to <60 mg m^{-2} in forest streams in the South Island (Friberg *et al.* 1997). At our Mangaoporo study sites, both gravels and cobbles supported only low levels of algal epilithon, probably because of bed movement, flaking of soft-rock surfaces and / or sandblasting by fine sediments. The expected effect on epilithon of light level differences between land-uses therefore was masked in the Mangaoporo streams.

Channel or substratum stability can also affect diversity of invertebrate communities (Death & Winterbourn 1995; Townsend *et al.* 1997b). At our study sites, invertebrate community composition varied along a stability gradient and taxa richness was correlated with stream stability. Te Arai streams were generally more stable than Mangaoporo streams and had correspondingly higher taxa richness. Within geographic clusters, stream stability and taxa richness were greater in pine plantations than pasture. Invertebrate density and taxa richness were significantly correlated with both stability and chl-*a*, however, we could not distinguish whether the physical effect of instability, or its effect on epilithon, was causing change in invertebrate communities. Both resource availability and disturbance may control invertebrate taxa richness, with disturbance acting by removing taxa and resetting the colonisation process (Death 2002). Because epilithic biomass, invertebrate density, taxa richness, and community composition of our study streams all differed according to stream stability, the geographic clustering of streams had a stronger overall effect than land-use. Friberg *et al.* (1997) also concluded that invertebrate communities and algal biomass differed more with geographical location than land-use, a finding they attributed largely to differences in bed stability of pine and beech forest streams east and west of the Southern Alps (regions of dramatically different rainfall regimes) in the South Island of New Zealand. Similarly, Horrox (1998) found that invertebrate community composition was more strongly related to substrate type (soft sedimentary versus hard rock substrata) than to land-use in streams of the central North Island.

Measures of stream health based on %EPT and QMCI, differed between forested and pasture streams, suggesting that riparian shade, and consequently water temperature, can have an important influence on sensitive invertebrate species in these streams. Native forest and pine plantation streams in both clusters were generally more similar to each other than to pasture streams, consistent with the results of a number of other land-use studies (Harding & Winterbourn 1995; Collier *et al.* 1997; Quinn *et al.* 1997). Both %EPT and QMCI were significantly and negatively correlated with water temperature, and invertebrate communities at our sites were separated along gradients of maximum daily water temperature in addition to stability and Chl-*a* in the CCA. Quinn *et al.* (1997) identified shade as the most important factor differentiating invertebrate community structure between pasture and forest streams in the Waikato and Friberg *et al.* (1997) concluded that difference in forest type has little effect on the benthic communities of the South Island streams they studied. Water temperature was expected to be an important factor as many stream invertebrates are sensitive to temperatures above 20 °C (Quinn *et al.* 1994; Cox & Rutherford 2000) and are often unable to survive in pastoral streams with high water temperatures or high air temperatures in the riparian zone (Collier *et al.* 1997; Quinn *et al.* 1997). In the pasture streams of both East Coast catchments, temperatures measured during February at each site were frequently in excess of 20 °C, and therefore likely to have had an impact on invertebrate communities, whereas the interquartile range of temperature data for all pine and native forest streams was below 20 °C.

Conclusion

Pasture streams draining soft rock terrain in both our study catchments in the Gisborne District had degraded water quality and invertebrate communities. Reforestation of pasture with pine on soft rock terrain resulted in stream habitat and water quality similar to native forest, at least within the time span (typically 28 years) of the first tree crop and prior to harvesting. Stream stability had a dominant influence on epilithon biomass and benthic invertebrate communities in our study. There was evidence that these pine plantation streams have greater stability and lower water temperature than the pastoral streams, resulting in improved stream ecological health. However, in large streams, bed instability may occur even under pine afforestation affecting invertebrate communities, and high sediment yields and turbidity are likely to persist where deep-seated geological disturbance has occurred. Furthermore, if pine trees are harvested up to the stream edge we would expect to find increases in water temperature and destabilisation of stream banks, with consequent impacts on water quality and stream health.

Acknowledgements

We are grateful to Noel Trustrum, Chris Phillips and Breck Bowden from Landcare Research for commissioning the work that was funded by the Foundation for Research, Science, & Technology contract C09X0214. The research was first suggested by Chris Ward (DOC, Gisborne). Thanks to NIWA's Hamilton chemistry laboratory for water quality analyses. We are especially grateful to Mark Meleason for performing the statistical analyses.

References

APHA (1998). *Standard methods for the examination of water and wastewater. 20th edition.* American Public Health Association, American Water Works Association, Water Pollution Control Federation. Washington.

Biggs, B.J.F. (2000). *New Zealand periphyton guidelines: detecting, monitoring and managing enrichment of streams.* Ministry for the Environment, Wellington.

Brinkhurst, R.O. (1971). The aquatic Oligochaeta known from Australia, New Zealand, Tasmania, and the adjacent islands. *University of Queensland papers, Department of Zoology* 3: 99-128.

Collier, K.J. (1992). *Assessing river stability: use of the Pfankuch method.* Science and Research Internal Report 131, Department of Conservation, Wellington.

Collier, K.J., Smith, B.J. & Baillie, B. (1997). Summer light-trap catches of adult Trichoptera in hill-country catchments of contrasting land-use, Waikato, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 31: 623-634.

Collins, R.P. (2003). Relationship between streamwater *E. coli* concentrations and environmental factors in New Zealand. Proceedings of the IWA Diffuse Pollution Conference, Dublin, Eire, September 2003. pp. 176-180.

Covert, T.C., Rice, E.W., Johnson, S.A., Berman, D., Johnson, C. & Mason, P.J. (1992). Comparing defined-substrate coliform tests for the detection of *Escherichia coli* in water.

Journal of the American Water Works Association May 1992: 98-104.

Cox, T.J. & Rutherford, J.C. (2000). Thermal tolerances of two stream invertebrates exposed to diurnally varying temperature. *New Zealand Journal of Marine and Freshwater Research* 34: 203-208.

Dall, P.C. (1979). A sampling technique for littoral stone-dwelling organisms. *Oikos* 33: 106-112.

Davies-Colley, R.J. (1988). Measuring water clarity with a black disc. *Limnology and Oceanography* 33: 616-623.

Davies-Colley, R.J. & Close, M.E. (1990). Water colour and clarity of New Zealand rivers under baseflow conditions. *New Zealand Journal of Marine and Freshwater Research* 24: 357-366.

Davies-Colley, R.J. & Payne, G.W. (1998). Measuring stream shade. *Journal of the North American Benthological Society* 17: 250-260.

Davies-Colley, R.J. & Stroud, M.J. (1995). *Water quality degradation by pastoral agriculture in the Whanganui River catchment.* NIWA Consultancy Report DOC050/1.

Davies-Colley, R.J. & Vant, W.N. (1987. Absorption of light by yellow substance in freshwater lakes. *Limnology and Oceanography* 32: 416-425.

Death, R.G. (2002). Predicting invertebrate diversity from disturbance regimes in forest streams. *Oikos* 97: 18-30.

Death, R.G. & Winterbourn, M.J. (1995). Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. *Ecology* 76: 1446-1460.

DeRose, R.C., Trustrum, N.A. & Blaschke, P.M. (1993. Post-deforestation soil loss from steepland hillslopes in Taranaki, New Zealand. *Earth Surface Processes and Landforms* 18: 131-144.

Eckner, K.F. (1998). Comparison of membrane filtration and multiple-tube fermentation by the Colilert and Enterolert methods for detection of waterborne coliform bacteria, *Escherichia coli*, and enterococci used in drinking and bathing water quality monitoring in southern Sweden. *Applied and Environmental Microbiology* 64: 3079-3083.

Friberg, N., Winterbourn, M.J., Shearer, K.A. & Larsen, S.E. (1997). Benthic communities of forest streams in the South Island, New Zealand: effects of forest type and location. *Archiv für Hydrobiologie* 138: 289-306.

Harding, J.S. & Winterbourn, M.J. (1995). Effects of contrasting land use on physico-chemical conditions and benthic assemblages of streams in a Canterbury (South Island, New Zealand) river system. *New Zealand Journal of Marine and Freshwater Research* 29: 479- 492.

Harding, J.S., Young, R.G., Hayes, J.W., Shearer, K.A. & Stark, J.D. (1999). Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology* 42: 345-357.

Hicks, D.M., Gomez, B. & Trustrum, N.A. (2000). Erosion thresholds and suspended sediment yields, Waipaoa River Basin, New Zealand. *Water Resources Research* 36: 1129-1142.

Hintze, J.L. (1998). *NCSS 2000 Quick start and self help manual.* Number Cruncher Statistical Systems, Kaysville, Utah.

Horrox, J.V. (1998). Benthic communities of the Whanganui River catchment: the effects of land use and geology. M.Sc. thesis, Massey University, Palmerston North.

Jowett, I.G. & Richardson, J. (1990). Microhabitat preferences of benthic invertebrates in a New Zealand river and the development of in-stream flow-habitat models for *Deleatidium* spp. *New Zealand Journal of Marine and Freshwater Research* 24: 19-30.

Kirk, J.T.O. (1976). Yellow substance (Gelbstoff) and its contribution to the attenuation of photosynthetically active radiation in some inland and coastal south-eastern Australian waters. *Australian Journal of Marine and Freshwater Research* 27: 61-71.

Marden, M. & Rowan, D. (1993). Protective value of vegetation on Tertiary terrain before and during Cyclone Bola, East Coast, North Island, New Zealand. *New Zealand Journal of Forestry Science* 23: 255-263.

Mazengarb, C. & Speden, I.G. (2000). Geology of the Raukumara area. Institute of Geological & Nuclear Sciences 1: 250 000 geological map 6. 1 sheet + 60 pp. Lower Hutt.

Ministry for the Environment (1992). *Water quality guidelines No. 1: Guidelines for the control of undesirable biological growths in water.* Ministry for the Environment, Wellington.

Ministry for the Environment (1994). *Water quality guidelines No. 2: Guidelines for the management of water colour and clarity.* Ministry for the Environment. Wellington.

Page, M.J., Trustrum, N.A. & DeRose, R.C. (1994). A high resolution record of storm-induced erosion from lake sediments, New Zealand. *Journal of Paleolimnology* 11: 333-348.

Parkyn, S.M. & Wilcock, R.J. (2004). Impacts of agricultural land use. In *Freshwaters of New Zealand* (eds J. Harding, P. Mosley, C. Pearson, & B. Sorrell) pp. 34.1-34.16. New Zealand Hydrological and Limnological Societies, Christchurch.

Pfankuch, D.J. (1975). *Stream reach inventory and channel stability evaluation.* U.S.D.A. Forest Service, Region 1, Missoula, Montana.

Quinn, J.M. & Hickey, C.W. (1990). Magnitude of effects of substrate particle size, recent flooding and catchment development on benthic invertebrates in 88 New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research* 24: 411-427.

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., Steele, G.L., Hickey, C. & Vickers, M.L. (1994). Upper thermal tolerances of twelve common New Zealand stream invertebrate species. *New Zealand Journal of Marine and Freshwater Research* 28: 391-397

Quinn, J.M. & Stroud, M.J. (2002). Water quality and sediment and nutrient export from New Zealand hill-land catchments of contrasting land use. *New Zealand Journal of Marine and Freshwater Research* 36: 409-429.

Richardson, J. & Jowett, I.G. (2002). Effects of sediment on fish communities in East Cape streams, North Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 36: 431-442.

Rutherford, J.C., Davies-Colley, R.J., Quinn, J.M., Stroud, M.J. & Cooper, A B. (1999). *Stream shade: towards a restoration strategy.* Miscellaneous

Report, Department of Conservation. Wellington.

Smith, D.G., Davies-Colley, R.J., Knoeff, J. & Slot, G.W.J. (1997). Optical characteristics of New Zealand rivers in relation to flow. *Journal of the American Water Resources Association* 33: 301-312.

Smith, D.G. & Maasdam, R. (1994). New Zealand's National River Water Quality Network 1. Design and physico-chemical characterisation. *New Zealand Journal of Marine and Freshwater Research* 28: 19-25.

Smith, C.M., Wilcock, R.J., Vant, W.N., Smith, D.G. & Cooper, A.B. (1993). *Towards sustainable agriculture: freshwater quality in New Zealand and the influence of agriculture.* Ministry of Agriculture and Fisheries Technical Policy Paper No. 93/10, Wellington.

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.

Taylor, N.H. (compiler) (1967). *Wise land use and community development: report of the technical committee of inquiry into the problems of the Poverty Bay - East Cape district of New Zealand.* Ministry of Works and Development, Wellington.

Townsend, C.R., Arbuckle, C.J., Crowl, T.A. & Scarsbrook, M.R. (1997a). 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.

Townsend, C.R., Scarsbrook, M.R. & Dolédec, S. (1997b). Quantifying disturbance in streams: alternative measures of disturbance in relation to macroinvertebrate species traits and species richness. *Journal of the North American Benthological Society* 16: 531-544.

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.

Wolman, M.G. (1954). A method of sampling coarse river-bed material. *American Geophysical Union Transactions* 35: 951-956.