

APPLIED ISSUES

# Scale and the detection of land-use effects on morphology, vegetation and macroinvertebrate communities of grassland streams

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## SUMMARY

1. Land-use studies are challenging because of the difficulty of finding catchments that can be used as replicates and because land-use effects may be obscured by sources of variance acting over spatial scales smaller than the catchment. To determine the extent to which land-use effects on stream ecosystems are scale dependent, we designed a whole-catchment study of six matched pairs (pasture versus native tussock) of second-order stream catchments, taking replicate samples from replicate bedforms (pools and riffles) in each stream.

2. Pasture streams had a smaller representation of endemic riparian plant species, particularly tussock grasses, higher bank erosion, a somewhat deeper layer of fine sediment, lower water velocities in riffles, less moss cover and higher macroinvertebrate biodiversity. At the bedform scale, suspendable inorganic sediment (SIS) was higher in pools than riffles and in pasture streams there was a negative relationship between SIS and the percentage of the bed free of overhanging vegetation. Differences between stream reaches (including any interactions between land use and stream pair) were significant for SIS, substrate depth and characteristics of riparian vegetation. There were also significant differences between replicate bedforms in the same stream reaches in percentage exotic species in overhanging vegetation, percentage moss cover, QMCI (Quantitative Macroinvertebrate Community Index – a macroinvertebrate-based stream health index) and macroinvertebrate density.

3. Significant differences among stream reaches and among replicate bedform units within the same reach, as well as interactions between these spatial units and land-use effects, are neither trivial nor 'noise' but represent real differences among spatial units that typically are unaccounted for in stream studies. Our multi-scale study design, accompanied by an investigation of the explanatory power of different factors operating at different scales, provides an improved understanding of variability in nature.

*Keywords:* bedform, catchment, native tussock, pasture, river geomorphology

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## Introduction

Land use has a major influence on community composition in streams in New Zealand (Quinn *et al.*, 1997; Riley *et al.*, 2003) and elsewhere (Richards, Johnson & Host, 1996; Sponseller, Benfield & Valett, 2001). Conversion of a catchment from one land use to

another may influence stream ecosystems via changes to nutrient loading (Allan, Erickson & Fay, 1997), solar energy flux (Hicks, 1997), hydrology (Davies-Colley, 1997), sediment inputs (Zweig & Rabeni, 2001), organic matter inputs (Hicks, 1997) and decomposition rates (Niyogi, Simon & Townsend, 2003). Such changes are often profound, particularly when forest is converted to pasture (e.g. Hicks, 1997; Wang *et al.*, 1997), but when native grassland is converted to pasture the changes may be more subtle (Townsend & Riley, 1999; Riley *et al.*, 2003).

Land-use studies provide particular challenges because they need to be conducted at large, landscape scales, incorporating whole or parts of catchments devoted to particular uses. Such units may be difficult to find and even more difficult to replicate in a rigorous manner. When the aim is to detect catchment-level patterns, a further problem arises if survey designs do not take account of small-scale variability. The multi-scale nature of ecological processes is certainly recognised by freshwater researchers (reviews: Allan & Johnson, 1997; Gergel *et al.*, 2002) and some have applied sophisticated analytical tools such as geographical information system to land-use data as a way of examining this issue (e.g. Hunsaker & Levine, 1995). Nevertheless, very few studies apply a similar level of sophistication to the survey design used to collect in-stream data. The commonest sampling design for water chemistry, algae and macroinvertebrates continues to be one site (or 'representative reach') of a 10–50 m length per river or catchment studied (e.g. Richards *et al.*, 1996, 1997; Johnson *et al.*, 1997; Harding *et al.*, 1998; Sponseller & Benfield, 2001; Sponseller *et al.*, 2001). The limitations of using individual sites to 'represent' larger spatial units, and the associated danger of arriving at faulty conclusions as a result, were clear from research conducted over a decade ago (e.g. Corkum, 1991, 1992; Downes, Lake & Schreiber, 1993), but seem to have had little impact on traditional, limnological sampling procedures.

The problem of using supposedly 'representative sites' can be exacerbated as well if there are poor selection procedures (e.g. when sites are picked arbitrarily and not randomly; Downes *et al.*, 2002). Finally, comparisons between sites may be flawed if stream lengths are selected without regard to standardising their geomorphological nature. Habitat units, such as riffles, plunge pools, rock faces and debris

dam pools, exist as a function of stream form and process. They are dependent on position in the landscape and hydrological regime (affecting sediment delivery and channel conditions), and are the product of regional climate, geology and vegetation (Allan & Johnson, 1997). The macroinvertebrate communities associated with habitat units depend on both the physical stream environment and on natural and human-induced factors that influence physical form and process (Townsend *et al.*, 2003).

In this study, we illustrate an alternative design to the standard one site/catchment approach to examining land-use effects. Our design had several scales of replication, which allowed us to test hypotheses concerning interactions between factors operating at different scales and to consider spatial variation among the units used to form replicates for tests. We examined six carefully matched pairs of second-order headwater streams, one of which flowed through native tussock grassland and the other through grassland improved for pasture. Streams were paired to match their geomorphological character, and we sampled different bedforms (pools and riffles). We tested the effects of land-use conversion on stream morphology, vegetation (overhanging, riparian vegetation and instream moss) and macroinvertebrate community composition. Conversion to pasture could result in trampling of stream banks by livestock and/or riparian changes from native tussock to pastoral grasses that do not have the same rooting strength or morphology. These hypothesised reductions in bank stability and subsequent erosion were predicted to create increased local supply of sediment, an increase in sediment stored in the channel, with attendant changes to the biota. These changes should be manifest at both catchment and bedform scales. If so, we expect to see significant land use and/or land use  $\times$  bedform interaction terms for our dependent variables.

Our other suite of hypotheses concerns the sources and significance of spatial variation. In each stream we took replicate samples from replicate pools and riffles. If there is significant spatial variability among scales smaller than that of whole catchments, we expect to see significant differences between stream pairs, between streams in the same land-use category or between replicate bedform units in the same stream. If variables are linked, then we expect them to show similar variability at different spatial scales.

## Methods

### Study sites

Our study catchments (Table 1) were in the Taieri and Waipori drainages of the Otago Province in the South Island of New Zealand. The pre-European condition of this area was native tussock grasslands on rolling topography underlain by Otago schist. Extensive grazing by sheep began 150 years ago and in the last 60 years some areas have been developed as 'improved' pastures through ploughing, sowing to exotic European grasses and applications with superphosphate and clover seed ('top-dressing'). Because of the mixed land ownership in this area, this is one of the few regions of Otago where pastoral and undeveloped native tussock grassland may still be found in close proximity at similar altitudes.

We used second-order streams in catchments that fell into two land-use categories: (i) relatively unmodified native tussock grassland catchments that have not been grazed for at least 8 years and (ii) catchments that have been converted to predominantly exotic pastoral grassland and grazed intensively by sheep and/or cattle. Although both native and exotic species were represented in both tussock and pasture sites, the riparian vegetation in tussock sites had a higher representation of native grasses [particularly *Chionochloa rigida* (Raoul) Zotov] together with native rushes [*Carex flaviformis* Nelmes, *Isolepis caligenis* (V. Cook) Soják, *Juncus gregiflorus* L. Johnson], herbs [*Galium perpusillum* (Hook. f) Allan, *Helichrysum bellidioides* (G. Forst.) Willd, *Hydrocotyle novae-zeelandiae* DC, *Oreomyrrhis ramosa* Hook. f., *Oxalis magellanica*

G. Forst., *Pratia angulata* (G. Forst.) Hook.f, *Ranunculus cheesemanii* Kirk, *Viola cunninghamii* Hook. f.] and shrubs [*Gaultheria macrostigma* (Colenso) D.J. Middleton, *Hebe* spp.]. Exotic pastures were characterised by exotic grasses (*Agrostis capillaris* L., *Anthoxanthum odoratum* L.) and herbs (*Cerastium fontanum* Baumg, *Hieracium pilosella* L., *Trifolium repens* L.).

### Physical variables

Sampling was conducted during several visits over a 3-month period from December 1998 to March 1999. During this summer period, precipitation was infrequent and low.

We grouped the 12 streams into six pairs (one tussock, one pasture) first on the basis of fish presence (some streams contain only introduced brown trout, *Salmo trutta* L., and others only native *Galaxias* spp.; Townsend, 2003). Stream pairs were then matched to control, as far as possible, for differences in catchment area, altitude, gradient, sinuosity, the ratio of bankfull width to bankfull depth, 50 year floodplain width and valley width (Tables 1 and 2). These physical variables were not expected to change substantially with land use.

Catchment area was calculated using 1 : 27 500 digitised aerial photographs (1996) and altitude was derived from 1 : 50 000 topographic maps. Reach gradient was measured using a TOPCON autolevel (Topcon Europe, Ijssel, The Netherlands) along the reach for a distance of about 30 channel widths or until there was a significant change in geomorphology, whichever was greater. Sinuosity was calculated by

**Table 1** Physical variables for six pairs of streams

Study streams	New Zealand map reference	Drainage area (km <sup>2</sup> )	Altitude (m)	Channel slope (%)	Sinuosity (m/m)	Bankfull width: bankfull depth	50-year floodplain width (m)	Valley width (m)
Tussock-1	H44:650883	1.59	650	2.5	1.07	2.3	1.2	6
Pasture-1	H44:681866	0.73	580	3.2	1.18	2.5	4.5	4.6
Tussock-2	H44:678923	1.06	600	3.9	1.02	3.0	1.5	20
Pasture-2	H44:656877	1.32	580	4.9	1.03	7.0	2.3	7.3
Tussock-3	H44:645912	1.03	700	4.1	1.02	2.9	2.9	23
Pasture-3	H44:659864	0.64	520	4.5	1.05	3.5	2.8	9
Tussock-4	H44:687943	1.05	660	2.5	1.1	6.3	2.5	3.4
Pasture-4	H44:714912	1.95	520	2.6	1.39	4.2	1	18
Tussock-5	H44:550810	0.64	520	4.3	1.04	4.6	1.3	5.2
Pasture-5	H44:618790	0.94	400	6.4	1.04	5.0	2.1	8
Tussock-6	H44:649936	1.21	540	2.6	1.31	2.1	1.8	23
Pasture-6	H44:587816	0.68	450	2.5	1.28	3.3	13.2	21.7

**Table 2** Definitions and units of physical variables

Variable	Definition	Units
Catchment area	Area of catchment area above sampled reach	km <sup>2</sup>
Gradient	Slope of the reach measured over 20 channel widths	m m <sup>-1</sup>
Valley width	Width of the valley where side slopes steeper	m
Floodplain width	50 year floodplain as defined by width at twice the bankfull depth	m
Altitude	Height above sea level as estimated from 1 : 50000 map	m
Sinuosity	Stream length/valley length in the valley which contains the stream	m m <sup>-1</sup>
Bankfull width	Width of the active channel at the point at which it spills onto a floodplain	m
Bankfull depth	Average depth of the channel at bankfull width	m
Erosion	Percentage of right and left banks that have exposed mineral soil	%
Bank height	Distance from streambed to floodplain or terrace	m
Bank angle	Angle between streambed and stream bank	Degrees
wtrdep	Depth of water at Surber	m
vel2	Velocity at 0.2 × total water depth	m s <sup>-1</sup>
vel4	Velocity at 0.4 × total water depth	m s <sup>-1</sup>
vel8	Velocity at 0.8 × total water depth	m s <sup>-1</sup>
froude	Average of vel2,vel4,vel8 divided by square root of gravity × water depth	Dimensionless
subdep	Depth of substrate at Surber as measured by metal rod pounded through loose, fine-grained material to impermeable layer	m
subtyp	Category (midpoint) of substrate size (e.g. wentworth scale sand is 0.062–2 mm and code = 0.25) at Surber site	mm
sis	Suspendable inorganic sediment at Surber site; resuspension of material within 14.6 cm tube as per Quinn <i>et al.</i> (1997)	g m <sup>-2</sup>
native	Percentage of vegetation species which overhang Surber site (0.5 m <sup>2</sup> ) that are native	%
exotic	Percentage of vegetation species which overhang Surber site (0.5 m <sup>2</sup> ) that are exotic	%
clsub	Percentage of 0.5 m <sup>2</sup> area within Surber site that is clear	%
moss	Percentage of 0.5 m <sup>2</sup> area within Surber site that has moss	%
area exotic	Percentage total area within plots covered with exotic species	%
area endemic	Percentage total area within plots covered with endemic, or native, species	%

measuring with a tape measure the actual course of the stream and dividing this distance by the straight-line distance from the top to the bottom of the reach along the valley axis. Bankfull widths and depths were measured at the top, middle and bottom of each pool and riffle within the reach. The 50-year floodplain was determined by multiplying by two the maximum bankfull depth at eight locations along the reach and extending a line at that height until the bank was intercepted (Dunne & Leopold, 1978; Rosgen, 1996). Valley width was measured at the point where steep valley slopes intercepted either the floodplain or, in two cases, a minor terrace (Tussock-2, Tussock-6).

Pools and riffles were defined according to water surface turbulence and speed of flow during the low flow sampling period (pools: slow, flat water; riffles: water flowing quickly over steeper substrate with an agitated, turbulent water surface). All bedforms (i.e. pools and riffles) along the stream reach (approximately 20 channel widths) were classified as pools and riffles (chutes and cascades, which are steeper

and faster than a traditional 'riffle', were not considered for sampling). Two pools and two riffles were randomly selected as sampling sites.

We measured bank erosion, bank height and bank angle ( $n = 16-20$ ) along the length of the reach. Bank erosion was estimated as the linear extent of exposed soils contributing directly to the stream as a percentage of the total length of both banks. Bank height was measured from the base to the top of a vertical bank at a point close to and directly above the active channel. If the floodplain was at the same level as the channel, bank height was recorded as near zero. Bank angle was measured using a protractor whose base was flush against the stream bottom (angles  $<90^\circ$  indicate undercut bank,  $90^\circ$  indicates a vertical bank, angles  $>90^\circ$  indicate a bank that slopes away from the stream).

The following additional physical variables were measured in each stream at two randomly selected locations within each pool and riffle sampled, and at the Surber site (see below). Velocity was measured, using a Marsh-McBirney Flomate instrument

(Marsh-McBirney, Frederick, MD, U.S.A.), at 0.2, 0.4 and 0.8 times the total depth. Substrate depth was measured by forcing a metal probe through soft, permeable sands or gravels until an impermeable layer was reached. Substrate was classified according to the Wentworth scale (Vanoni, 1977), using the midpoint of the class. For finer material that could be easily entrained in the water column, we used a modification of the suspendable sediment technique employed by Quinn *et al.* (1997). We placed a 14.6 cm diameter tube fitted with a foam plastic collar on the streambed to form a temporary still-water pool. The surface substrate to a depth of 5 cm was disturbed using a metal ruler for 10 s to suspend as much material as possible. Replicate water samples in the tube were taken, and the depth of water in the tube was measured. Water samples were analysed for suspended sediment by filtering through pre-combusted glass fibre filters, drying at 60 °C, weighing, incinerating at 550 °C for 2 h and reweighing. The masses of inorganic and organic material were determined for each sample and converted to mass of suspended material per area of streambed. Table 2 summarises the physical variables and their definitions.

#### Riparian vegetation

Vegetation overhanging a 0.0625 m<sup>2</sup> quadrat at each sampling location was classified as native or exotic and the percentage cover of each was estimated by eye. The percentage of substrate that was clear of overhanging vegetation and the percentage of the bed that was covered by moss were also estimated.

Six 10 × 5 m plots in the riparian zone were randomly selected along the stream reaches and the percentage of all native species and specifically of native tussock grass were estimated. Areas were evenly distributed between right and left banks. Species were identified to species level where possible.

#### Macroinvertebrates

Macroinvertebrates were collected using a Surber sampler (0.0625 m<sup>2</sup> area, 250 µm mesh, sampling to a depth of 5 cm in the substrate) from each of the two locations in each bedform sampling site. Samples were placed in plastic bags, sealed and frozen upon return to the laboratory for analysis at a later date.

After defrosting, samples were sorted by hand and invertebrates were identified to species or genus using the keys of Winterbourn & Gregson (1989) and Chapman & Lewis (1976). Taxon richness was calculated as the number of invertebrate taxa at each site. The Quantitative Macroinvertebrate Community Index (QMCI; a stream health index widely used in New Zealand), which weights species abundance by species-specific indices of tolerance to organic pollution, was calculated according to Stark (1993). We also calculated the percentage of the invertebrate community comprised of Ephemeroptera, Plecoptera and Trichoptera (EPT), based both on numbers and biomass. Such indices of stream health are used internationally (Plafkin *et al.*, 1989).

#### Statistical analysis

Data collected at the whole reach level (e.g. vegetation in the catchment, bank characteristics) were analysed as a randomised blocks design. Data collected to investigate the effects of land-use conversion were analysed using an unusual analysis of variance model (Table 3). The overall layout has some characteristics

**Table 3** The analysis of variance model used to analyse all survey data. Provided are the terms in the model, whether they are fixed or random, degrees of freedom, and the mean square (MS) term that supplies the denominator for the *F*-test. Some terms are tested over interaction terms because one (Pairs) is a random factor (see Kirk, 1995 for more information). The tests of land-use and land-use × bedform tell us whether characteristics of interest differ between pasture and tussock streams. The tests of pairs, stream reaches, P × B, B × R and replicate bedform units tell us about the sizes and sources of otherwise unexplained spatial variability in and among the study streams. There were 96 samples in total

Model term	Fixed or random	d.f.	Test versus:
Between plots			
Land use (L)	Fixed	1	MS <sub>R</sub>
Pairs (P)	Random	5	MS <sub>R</sub>
Stream reaches (R)		5	MS <sub>T</sub>
(=between-streams + land use × pairs)			
Within plots			
Bedform (B)	Fixed	1	MS <sub>PB</sub>
L × B		1	MS <sub>BR</sub>
P × B		5	MS <sub>BR</sub>
B × R		5	MS <sub>T</sub>
Replicate bedform units (T) <sub>w B × R</sub>	Random	24	MSE
Samples/residual error (E)	Random	48	

**Table 4** Analysis of variance of characteristics of the study streams, using the model given in Table 3

Source	MS	F	P	SA	MS	F	P	SA
	Percentage of substrate clear of vegetation (arcsine transformed)				Exotic vegetation			
Between plots								
Land use (L)	0.28	0.16	0.71	0.0	9985.80	3.82	0.11	20.6
Pairs (P)	0.56	0.32	0.88	0.0	2587.68	0.99	0.50	0.0
Stream reaches (R)	1.77	15.22	<b>&lt;0.001</b>	39.0	2614.19	6.29	<b>&lt;0.001</b>	36.8
Within plots								
Bedform (B)	0.20	1.08	0.35	0.1	499.14	0.95	0.37	0.0
L × B	0.02	0.02	0.89	0.0	1.24	0.003	0.96	0.0
P × B	0.19	0.29	0.90	0.0	523.88	1.14	0.44	1.1
B × R	0.66	5.72	<b>0.001</b>	25.8	458.96	1.10	0.38	1.5
Replicate bedform units <sub>w B × R</sub>	0.12	0.62	0.89	0.0	415.46	2.27	<b>0.01</b>	15.6
Residual	0.19			35.1	183.07			24.5
	Suspendable inorganic sediment (log transformed)				Percentage cover of moss (log transformed)			
Between plots								
Land use (L)	0.26	0.16	0.71	0.0	6.39	13.66	<b>0.01</b>	27.1
Pair (P)	0.25	0.15	0.97	0.0	1.26	2.69	0.15	10.9
Stream reaches (R)	1.70	13.22	<b>&lt;0.001</b>	44.4	0.47	1.40	0.26	3.6
Within plots								
Bedform (B)	2.94	20.03	<b>0.01</b>	13.2	0.42	0.72	0.44	0.0
L × B	0.21	0.50	0.51	0.0	0.19	0.43	0.54	0.0
P × B	0.15	0.35	0.86	0.0	0.58	1.31	0.39	3.8
B × R	0.42	3.26	<b>0.02</b>	16.4	0.45	1.33	0.29	6.0
Replicate bedform units <sub>w B × R</sub>	0.13	1.26	0.24	3.4	0.34	3.11	<b>&lt;0.001</b>	24.9
Residual	0.10			22.6	0.11			23.7
	Velocity (log transformed)				Froude number (log transformed)			
Between plots								
Land use (L)	0.19	3.63	0.11	5.0	0.25	3.27	0.13	4.7
Pair (P)	0.09	1.73	0.28	4.3	0.05	0.69	0.65	0.0
Stream reaches (R)	0.05	2.35	0.07	6.4	0.08	2.91	<b>0.03</b>	8.3
Within plots								
Bedform (B)	0.99	19.72	<b>0.01</b>	33.6	1.63	39.16	<b>0.002</b>	43.9
L × B	0.12	17.43	<b>0.01</b>	7.9	0.10	8.12	<b>0.04*</b>	5.0
P × B	0.05	7.24	<b>0.02</b>	8.6	0.04	3.28	0.11	5.0
B × R	0.01	0.31	0.90	0.0	0.01	0.49	0.78	0.0
Replicate bedform units <sub>w B × R</sub>	0.02	1.20	0.29	0.0	0.03	1.20	0.29	6.6
Residual	0.02			34.3	0.02			26.5
	Substrate depth (log transformed)							
Between plots								
Land use (L)	8.82	5.23	0.07	25.7				
Pairs (P)	0.56	0.33	0.88	0.0				
Stream reaches (R)	1.69	6.57	<b>0.001</b>	30.8				
Within plots								
Bedform (B)	0.77	16.93	<b>0.01</b>	2.6				
L × B	0.01	0.07	0.81	0.0				
P × B	0.05	0.41	0.83	0.0				
B × R	0.11	0.43	0.82	0.0				
Replicate bedform units <sub>w B × R</sub>	0.26	1.18	0.30	3.5				
Residual	0.22			37.4				

MS, mean square; F, F-statistic; P, probability; SA, strength of association. Probabilities <0.05 are in bold. Degrees of freedom and denominators for F-tests are reported in Table 3.

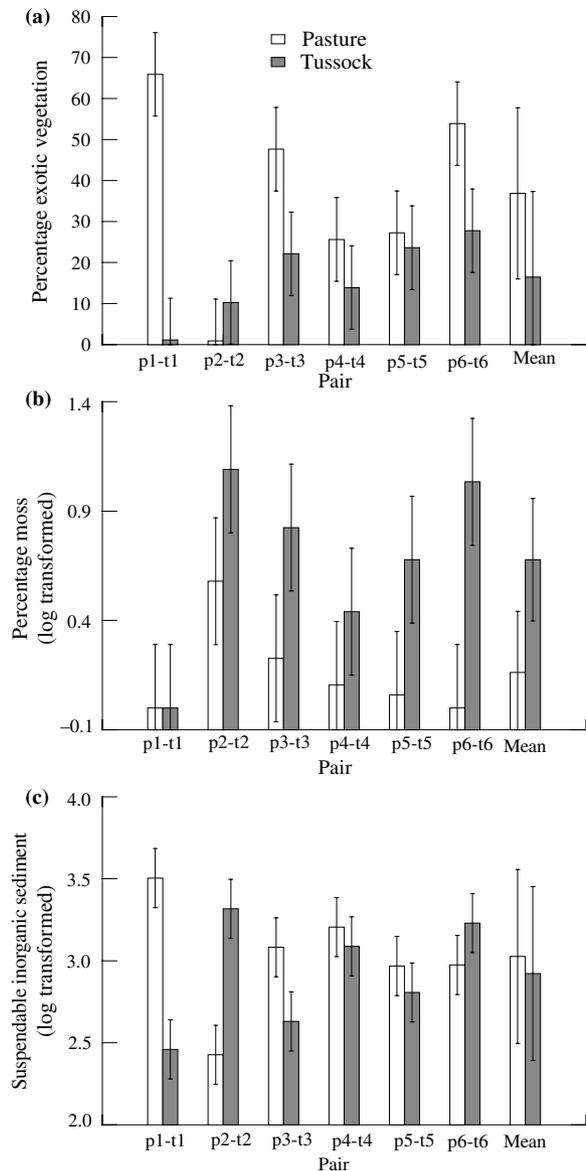
\*F-test = 5.16, P = 0.07 with outlier removed.

of a split-plot (Kirk, 1995). Split-plot designs were championed by agricultural researchers who used plots of land that were subdivided (or split) into subplots. The basic idea is that one or more experimental treatments are applied to whole plots, whereas others are applied to subplots. Often the effects of most interest are the latter and their interactions with plot-level treatments. Data analysis is relatively straightforward using analysis of variance, and there will be at least two error terms in the model (Kirk, 1995). These designs are ideal for exploring effects of land-use changes and their interactions with physical processes acting over different spatial scales. Thus, in our study, stream reaches are plots that form the replicates for effects of changes in land use (tussock versus pasture), changes that have been created at relatively large scales. Nevertheless, we expect some land-use effects (e.g. sediment density) to differ between bedforms within catchments. Consequently, we recognise smaller units (riffles and pools) as subplots that form the replicates for the effects of bedform and interactions of the latter with land use.

There are two somewhat unusual features to our split-plot design (Table 3). First, the between-plot effects are arranged as a randomised block design, because stream reaches in different land-use categories were very carefully paired for matching physical features and hence were not selected purely at random. This means that the model includes a blocks term (Pairs) and that the between-plots error term, Stream reaches, includes both differences between individual streams and any variance caused by land use  $\times$  pair interactions, which cannot be separately estimated by definition (Kirk, 1995). Secondly, there are replicate sub-plots [replicate bedform units (T)], and this has two implications for the analysis. First, there is a mean square term ( $MS_T$ ) that can form a test for stream reaches. The mean square for the latter ( $MS_R$ ) automatically confounds the effects of two sorts of variability as noted above, but the test allows us to ask whether such variance is collectively important. Secondly, the design does not have the usual, non-estimable interaction term (which in our design is  $B \times$  stream reaches) and hence the residual can be separately estimated (Kirk, 1995).

We also calculated the strength of association of each term in the model as a way of looking at effect sizes – the actual sizes of differences between (for example) sample means (Winer, Brown & Michels,

1991; Kirk, 1995). Strength of association is measured by the statistics  $\rho_I$  and  $\omega^2$  for random and fixed effects, respectively. These statistics estimate the proportion of variance explained by the term and are reported as a percentage of the total. These values tell us some-



**Fig. 1** The average of (a) percentage of over-hanging vegetation that was exotic, (b) percentage cover of moss and (c) suspendable inorganic sediment ( $\text{g m}^{-2}$  log transformed) in pairs of pasture (p) and tussock (t) streams (details are given Table 1) together with the overall means across all pairs of streams (bars labelled 'Mean'). Error bars are standard errors that have been generated from the appropriate MS term from the analysis of variance.

thing about the 'importance' of the term, because a high value for strength of association signals a strong effect in a way that an *F*-test, by itself, does not.

All data were analysed using SYSTAT version 9. During analysis, residuals were inspected and data transformed if variances were heterogeneous between cells of the design. Effects of any outlying values were tested by rerunning analyses without those observations and determining effects on *F*-tests. Any significant changes are reported in tables.

## Results

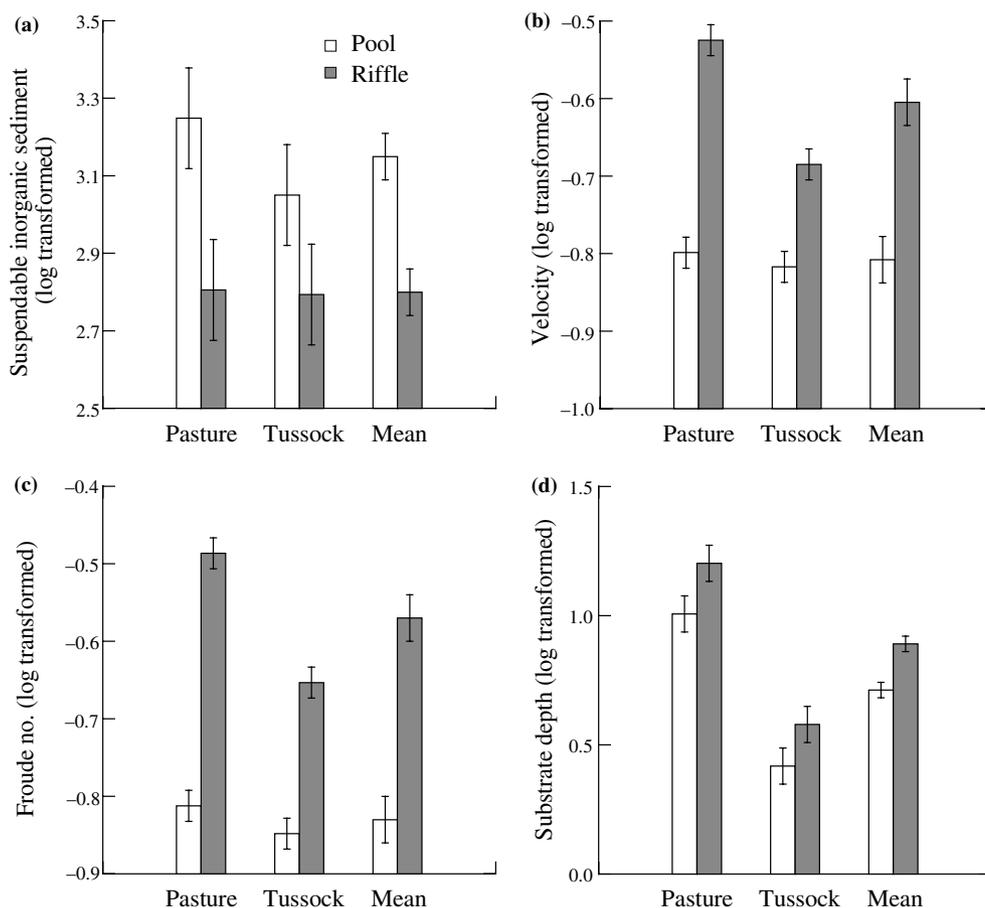
### Catchment-level differences

We report first on whether streams did indeed differ in the characteristics associated with conversion of land use from tussock to pasture. Compared with

tussock streams, pasture streams had a lower percentage of riparian area covered by endemic plant species compared with exotics (26.5 versus 56.7%;  $F_{1,5} = 26.7$ ,  $P = 0.004$ ) and less ground covered by tussocks (0.22 versus 0.92%;  $F_{1,5} = 11.2$ ,  $P = 0.02$ ). Pasture streams had significantly higher bank erosion (30.5% versus 5.5%,  $F_{1,5} = 12.91$ ,  $P = 0.02$ ); their banks were also lower and more likely to be vertical or, in some cases, to slope away from the stream, but these trends were not as strong (bank height: 0.28 versus 0.40 m,  $F_{1,5} = 3.9$ ,  $P = 0.11$ ; bank angle: 87.8° versus 64.8°,  $F_{1,5} = 5.9$ ,  $P = 0.06$ ).

### In-channel differences associated with land use

A number of in-channel characteristics showed differences between pasture and tussock streams. Streams in pasture had far less moss covering the substrate

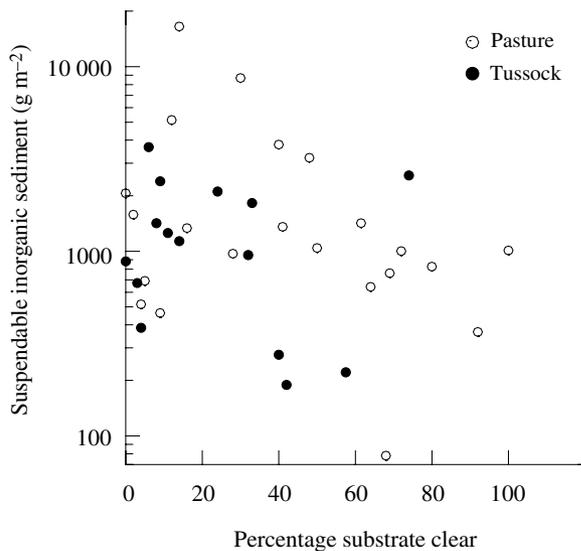


**Fig. 2** The average of (a) suspended inorganic sediment ( $\text{g m}^{-2}$ ), (b) velocities ( $\text{m s}^{-1}$ ), (c) Froude number and (d) substrate depths (cm) in pools and riffles in pasture and tussock streams and across all streams (bars labelled 'Mean'). All variables are plotted in log-transformed values. Error bars as for Fig 1.

(Table 4; Fig. 1). There were no strong differences in the degree to which overhanging plants on the bank shaded the substrate, but a somewhat higher % of such vegetation in pasture streams was of exotic origin (Fig. 1). Average water velocity in riffles (and also Froude number) was higher in pasture than tussock streams but there were no differences in average velocities in pools (Fig. 2). Pasture streams had somewhat deeper layers of sediment than tussock streams (Table 4; Fig. 2 – the trend is not quite statistically significant), but there were no strong differences between them in amounts of suspendable inorganic sediment (SIS). SIS varied considerably within pairs of streams (Fig. 2) but, as expected, was higher in pools than riffles (Fig. 2). This variability matched that of percentage Surber area free of overhanging vegetation (Table 4) and a scatterplot showed these two variables to be negatively associated at the bedform unit scale (Fig. 3). This association is significant for pasture streams (Pearson correlation  $r = -0.43$ ,  $P = 0.03$ ) but not tussock streams ( $r = -0.16$ ,  $P = 0.45$ ).

#### Macroinvertebrate variability associated with land use

Number of invertebrate taxa, EPT density and biomass all showed significantly higher values in pasture



**Fig. 3** Scatterplot of the amount of suspendable inorganic sediment ( $\text{g m}^{-2}$ ) versus the percentage of the substrate clear of overhanging vegetation for pasture (open circles) and tussock (closed circles) streams. Each point is the average of two Surber samples in one bedform unit.

compared with tussock streams (Table 5; Fig. 4), although  $F$ -tests for the latter two variables were affected by outliers and hence the results must be viewed with caution. QMCI scores were also higher in pasture than tussock streams, although the trend was not statistically significant (Table 5; Fig. 4). Overall densities of macroinvertebrates did not differ between pasture and tussock streams (Table 5; Fig. 4). Riffles, compared with pools, had higher numbers of taxa, greater densities of macroinvertebrates and greater EPT densities and biomasses; these effects were the same in pasture and tussock streams (Fig. 5). There was also an association between number of invertebrate taxa and percentage cover of moss at the bedform unit scale (Fig. 6) for both pasture (Pearson correlation  $r = 0.49$ ,  $P = 0.02$ ) and tussock ( $r = 0.55$ ,  $P = 0.005$ ) streams, although in both cases there is a great deal of scatter.

#### Spatial variation and strength of association

Some variables showed a remarkable degree of spatial variability (Tables 4 and 5). Differences between stream reaches (which includes any interactions between land-use type and pair) were significant for many variables; for example, stream reach differences explained an estimated 44% of variance in SIS, 31% of variability in substrate depth and over 35% of variation in characteristics of riparian vegetation. Several variables also showed significant bedform  $\times$  stream reach variation. Four variables showed significant differences between replicate bedform units within the same length of stream: QMCI, total densities of macroinvertebrates, percentage of overhanging vegetation that was exotic and percentage cover of moss. In contrast, the effect sizes of land use, bedform and interactions between them explained usually smaller percentages of variation, although land use explained 27% of variation in moss cover and a similar amount in substrate depth.

## Discussion

### Land-use effects

The results of this study lend further weight to the conclusion that the terrestrial components of pasture catchments have been strongly influenced by land management practices of ploughing, sowing of exotic

**Table 5** Analyses of variance in various measures of macroinvertebrate density and diversity. Abbreviations as in Table 4

Source	MS	F	P	SA	MS	F	P	SA
	No. of invertebrate taxa				Total density of invertebrate individuals (log-transformed)			
Between plots								
Land use (L)	234.38	9.04	<b>0.03</b>	18.3	0.11	0.14	0.72	0.0
Pairs (P)	33.28	1.28	0.40	1.9	0.76	0.97	0.51	0.0
Stream reaches (R)	25.93	2.26	0.08	7.6	0.79	3.33	<b>0.02</b>	22.1
Within plots								
Bedform (B)	150.00	14.35	<b>0.01</b>	12.2	2.65	16.46	<b>0.01</b>	16.7
L × B	10.67	0.39	0.56	0.0	0.35	1.02	0.36	0.1
P × B	10.45	0.38	0.84	0.0	0.16	0.47	0.78	0.0
B × R	27.47	2.40	0.07	16.9	0.34	1.44	0.25	8.0
Replicate bedform units <sub>w B × R</sub>	11.46	1.27	0.23	5.2	0.24	2.57	<b>0.003</b>	24.1
Residual	9.00			37.9	0.09			28.9
	Density of EPT (log-transformed)				Biomass of EPT (log-transformed)			
Between plots								
Land use (L)	4.11	8.71	<b>0.03*</b>	12.5	2.78	6.54	<b>0.05<sup>‡</sup></b>	11.3
Pairs (P)	0.93	1.96	0.24	4.8	0.86	2.02	0.23	6.3
Stream reaches (R)	0.47	1.80	0.15	4.3	0.42	1.90	0.13	5.7
Within plots								
Bedform (B)	5.21	8.69	<b>0.03</b>	15.9	1.98	7.24	<b>0.04<sup>§</sup></b>	8.2
L × B	1.04	1.31	0.30	1.7	2.00	5.84	0.06	15.8
P × B	0.60	0.75	0.62	0.0	0.27	0.80	0.59	0.0
B × R	0.79	3.03	<b>0.03<sup>†</sup></b>	21.9	0.34	1.53	0.22	6.9
Replicate bedform units <sub>w B × R</sub>	0.26	1.27	0.24	4.1	0.22	1.24	0.26	4.6
Residual	0.21			34.7	0.18			41.2
	QMCI							
Between plots								
Land use (L)	31.23	5.50	0.07	21.8				
Pairs (P)	1.70	0.30	0.89	0.0				
Stream reaches (R)	5.68	6.31	<b>0.001</b>	24.5				
Within plots								
Bedform (B)	9.56	2.97	0.15	5.4				
L × B	0.53	0.20	0.67	0.0				
P × B	3.22	1.23	0.41	3.1				
B × R	2.62	2.91	<b>0.03</b>	17.6				
Replicate bedform units <sub>w B × R</sub>	0.90	2.00	<b>0.02</b>	9.2				
Residual	0.45			18.4				

\*F-test is 5.00, P = 0.08 with outlier removed.

†F-test is 2.37, P = 0.07 with outlier removed.

‡F-test is 8.48, P = 0.03 with outlier removed.

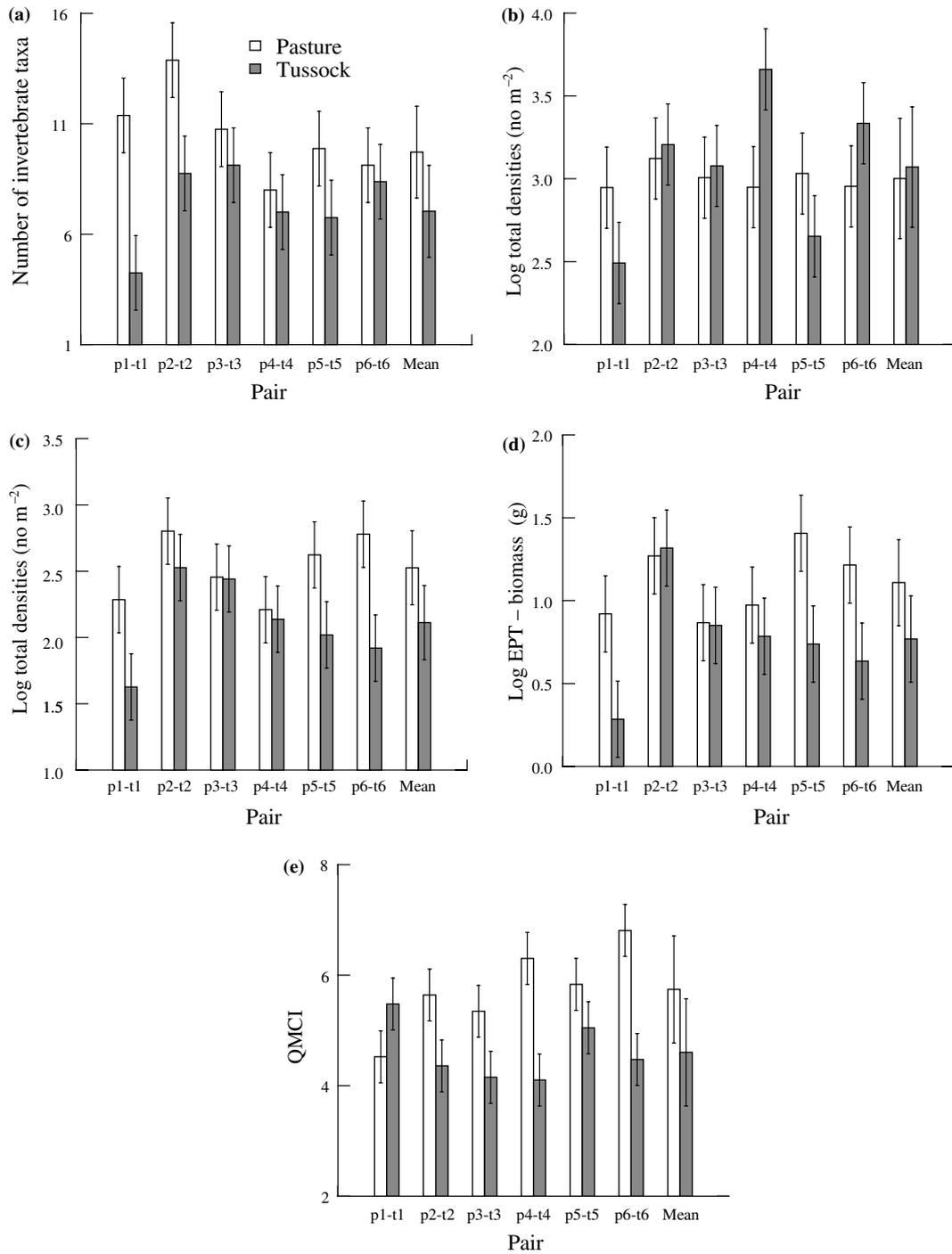
§F-test is 4.91, P = 0.08 with outlier removed.

species, fertilisation and high stocking rates (Riley *et al.*, 2003). Thus, tussock grasses have been eliminated from ploughed areas and grazing and trampling have reduced tussock canopy coverage in unploughed riparian margins of pasture sites. Increased bank erosion in pasture settings was associated with the loss of undercut banks in many instances and was responsible for somewhat deeper sediment on the streambed and a reduction in moss cover, partly because of the loss of clean bedrock surfaces that normally provide a stable substrate for moss (Riley *et al.*, 2003). Macroinvertebrate biodiversity

was generally higher in pasture than tussock streams. Riley *et al.* (2003) consider this to be a reflection of the potentially negative effects of sediment deposition being outweighed by a positive effect of higher nutrient levels on algal primary productivity fuelling more complex food webs (Townsend *et al.*, 1998).

#### Significance of spatial variation

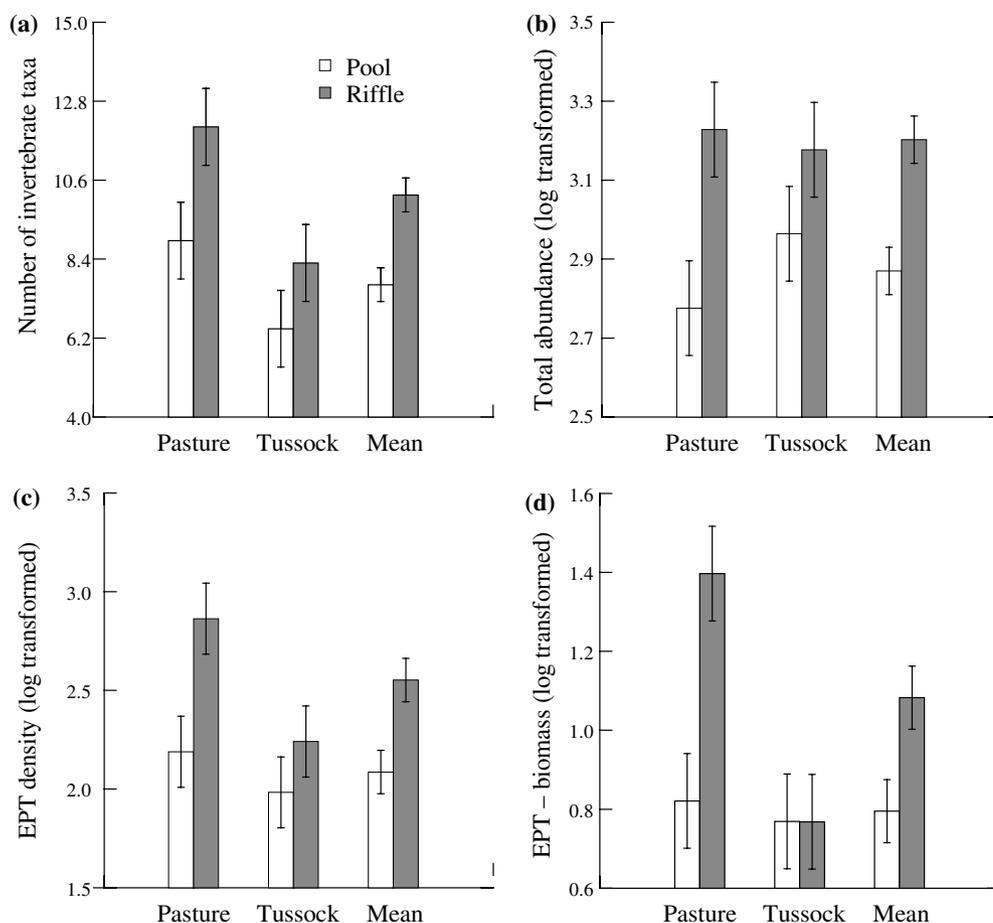
The primary aim of this study was not, however, simply to quantify land-use effects on streams and their margins. Rather our focus was on the question of



**Fig. 4** The average of (a) number of invertebrate taxa, (b) total densities of macroinvertebrates (no. m<sup>-2</sup>, log transformed), (c) densities (no m<sup>-2</sup> log transformed) and (d) biomasses (g, log transformed) of EPT taxa and (e) values of QMCI in pairs of pasture (p) and tussock (t) streams (details are given in Table 1) together with the overall means across all pairs of streams (bars labelled 'Mean'). Error bars as for Fig. 1.

variation among spatial units that are usually used to test hypotheses such as land use–stream health relationships. It is of particular significance that our

study design allowed the detection of frequent and large variation among commonly used spatial units, including those smaller than the catchment (or, more



**Fig. 5** The average of (a) number of invertebrate taxa, (b) total densities of macroinvertebrates and (c) densities and (d) biomasses of EPT taxa in pools and riffles in pasture and tussock streams and across all streams (bars labelled 'Mean'). All variables except number of invertebrate taxa are plotted in log-transformed values. Error bars as for Fig. 1.

specifically, the stream reach). We found significant differences among stream reaches and among replicate bedform units within the same reach, as well as interactions between these spatial units and land-use effects of interest. This variation is neither trivial nor 'noise': it represents real differences among spatial units that typically go unaccounted for in most stream studies. Its presence has several important implications.

First, the streams in this study were carefully chosen and paired for characteristics considered 'important'. Sampling procedures like this, where researchers choose streams to 'match' them for particular characteristics, are common in land-use studies. However, in contrast to perceptions underlying such experimental designs, we found large differences in some characteristics between paired stream reaches

(and/or interactions between individual streams and land use – we cannot isolate this source of variance). Another common presumption is that stream researchers can choose reaches that will 'represent' some (usually unspecified) much longer length of channel, a presumption that specifically assumes variation between sites along channels is trivial. This presumption (like the one above about 'matching streams') is rarely put to the test or even questioned, and, we suggest, is suspect for variables that cannot be directly observed and require samples (e.g. algae, macroinvertebrates). For example, we found that macroinvertebrate diversity and abundance responded to factors operating at quite small scales. There were also interactions between these small-scale processes with factors varying at larger scales (such as effects of human impacts), resulting in significant

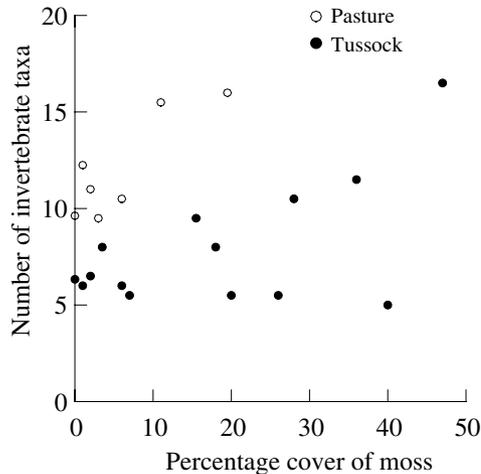


Fig. 6 Scatterplot of the number of invertebrate taxa versus the percentage of the substrate covered by moss for pasture (open circles) and tussock (closed circles) streams. Each point is the average of two Surber samples in one bedform unit.

differences among replicate riffles in the same stream reach. Such complex responses can only be teased out using survey designs that include tests for cross-scale interactions and that do not presume variability at scales smaller than that of catchments or whole rivers is trivial. Additionally, such designs provide a more detailed and sophisticated picture of human effects on the environment than simple (e.g. reference versus impacted sites) approaches.

It is worth considering further what happens when simple survey designs (e.g. one-way comparison between reference and impacted sites, where sites are used to 'represent' whole rivers or catchments) are applied to data where the sort of complex interactions and spatial variability described above are present. The obvious result is that much variability is unaccounted for, ends up in error terms, and hence compromises the ability of investigators to detect effects (given that the basis of statistical tests is to detect effects above and beyond variability among replicate units). A more insidious, and therefore worrying, effect is that investigators can reach faulty conclusions. When single sites (often riffles or sections thereof) are used to 'represent' whole streams (or whole catchments), any differences among sites will be ascribed to differences between whole streams (or whole catchments). This creates a potential for confusing between-site variation with between-stream variation, and, if our streams are any guide, the probability of doing that will be high. Consequently,

researchers might conclude that they have detected a difference among streams that was driven by land-use patterns, when in reality they have picked up a between-site pattern that happened to be coincident with land-use patterns. The converse finding, that no land-use effect was detected, is equally worrying. In both cases, researchers have no way of disentangling site-to-site variability from stream-to-stream variability because they do not have appropriate replication. Similar worries have been echoed in other studies (e.g. Manel, Buckton & Ormerod, 2000) working at far larger scales, overall, than the current study.

Finally, it should be noted that the sort of survey design used here is not necessarily more expensive or time-consuming than a simply designed survey (especially when the latter includes hundreds of dependent variables). The aim of any study should be to test clear, well-developed hypotheses and hence to expend effort and replication where it is needed. It is often presumed, for example, that quantifying macroinvertebrate diversity or abundance must always include separately enumerated Surber samples. In most human impact studies (and many others as well), variation between individual Surber samples is of no relevance because hypothesised effects act at larger scales. Surber samples add no degrees of freedom to pertinent tests, as should be apparent from Table 3. Replication of Surber samples ensures that densities of macroinvertebrates in units like riffles are well estimated, but analyses use the average of Surber samples, regardless of whether two or 20 samples have been taken. If the requirement to take many Surber samples at every site is relaxed, sampling effort can be targeted at those scales (e.g. riffles, sites, streams and larger catchments) that are most relevant. We suggest that such an approach, especially where accompanied by investigation of the explanatory power (e.g. strength of association) of different factors operating at different scales, will result in better understanding of variability in nature.

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