Contrasting responses to catchment modification among a range of functional and structural indicators of river ecosystem health

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SUMMARY

1. The value of measuring ecosystem functions in regular monitoring programs is increasingly being recognized as a potent tool for assessing river health. We measured the response of ecosystem metabolism, organic matter decomposition and strength loss, and invertebrate community composition across a gradient of catchment impairment defined by upstream landuse stress in two New Zealand streams. This was done to determine if there were consistent responses among contrasting functional and structural indicators.
2. Rates of gross primary production (GPP) and ecosystem respiration (ER) ranged from 0.1-7.0 gO_2 m^{-2} day^{-1} and 0.34-16.5 gO_2 m^{-2} day^{-1}, respectively. Rates of GPP were variable across the landuse stress gradient, whereas ER increased linearly with the highest rates at the most impacted sites. P/R and Net Ecosystem Metabolism (NEM) indicated that sites at the low and high ends of the stress gradient were heterotrophic with respiration rates presumably relying on organic matter from upstream sources, adjacent land or point sources. Sites with moderate impairment were predominantly autotrophic.
3. Declines in the tensile strength of the cotton strips showed no response across part of the gradient, but a strong response among the most impaired sites. The rate of mass loss of wooden sticks (*Betula platyphylla* Sukaczev) changed from a linear response to a U-shaped response across the impairment gradient after water temperature compensation, whereas leaf breakdown at a subset of sites suggested a linear loss in mass per degree-day. Three macroinvertebrate metrics describing

the composition of the invertebrate community and its sensitivity to pollution showed similar linear inverse responses to the landuse stress gradient.

4. The first axis of a redundancy analysis indicated an association between landuse stress and various measures of water quality, and wooden stick mass loss, the invertebrate metric % EPT taxa, P/R and NEM, supporting the utility of these structural and functional metrics for assessing degree of landuse stress. The second axis was more strongly associated with catchment size, ER and GPP which suggests that these indicators were responding to differences in stream size.
5. Our results suggest that non-linear responses to catchment impairment need to be considered when interpreting measurements of ecosystem function. Functional indicators could be useful for detecting relatively subtle changes where the slope of the response curve is maximized and measurements at the low and high ends of the impairment gradient are roughly equivalent. Such responses may be particularly valuable for detecting early signs of degradation at high quality sites, allowing management responses to be initiated before the degradation becomes too advanced, or for detecting initial moves away from degraded states during the early stages of restoration. Close links between structural and functional indices of river health across an impairment gradient are not necessarily expected or desirable if the aim is to minimize redundancy among ecological indicators.

Key words: biotic indices, decomposition , ecosystem metabolism, organic matter , river health

Introduction

The health or integrity of river ecosystems throughout the world continues to be threatened by a wide range of anthropogenic factors such as intensification of land use and increasing demand for fresh water. To maintain or improve river ecosystem health, tools for assessing the current ecological state of river ecosystems are needed so the extent of degradation can be determined and the success of rehabilitation efforts measured. In the past, these tools have concentrated on ecosystem structure, but recently the value of incorporating measurements of ecosystem function into monitoring programs has increasingly been recognized (Bunn & Davies 2000; Gessner & Chauvet 2002; Carlisle & Clements 2005; Paul, Meyer & Couch 2006; Uehlinger 2006; Young, Matthaei & Townsend 2008). In particular, functional indicators may have utility for discriminating low levels of impairment which is often problematic using conventional structural indicators, and may also be able to detect initial moves away from degraded states required to demonstrate tangible improvements in ecosystem health during the early stages following restoration (Palmer *et al.* 2005). Functional measures can also provide a direct measure of valuable ecosystem services, which satisfy human needs for economic, social and health-related benefits in addition to the inherent value of ecosystem health (Rapport, Costanza & McMichael 1998).

There are a variety of ecosystem processes that could potentially be used as indicators of river ecosystem health. These include rates of nutrient uptake (Sabater *et al.* 2000; Hall & Tank 2003), benthic microbial respiration (Niyogi, Lewis & McKnight 2001; Hill, Herlihy & Kaufmann 2002), denitrification (Bernhardt, Hall & Likens 2002; Udy *et al.*

2006), fine particulate organic matter export (Wallace, Grubaugh & Whiles 1996), organic matter retention (Speaker, Moore & Gregory 1984; Quinn, Phillips & Parkyn 2007) and invertebrate production (Woodcock & Huryn 2007). However, some of these involve large effort or sophisticated and expensive techniques. Alternatively, rates of organic matter decomposition and ecosystem metabolism appear particularly suited as indicators since they respond to a range of physical and chemical stressors (Pascoal *et al.* 2003; Mulholland *et al.* 2001), are relatively inexpensive, and metabolism measurements at least are amenable to automation (Izagirre *et al.* 2008; Young *et al.* 2008). All river and stream ecosystems are fuelled by a combination of terrestrially derived organic material and autochthonous material produced in-stream. Thus, measurements of the rate of organic matter decay and ecosystem metabolism provide indications of the food-base of the ecosystem, and thus help determine the basis underlying its life-supporting capacity (Fisher & Likens 1973).

An important consideration with any indicator is what the measurements mean in terms of ecosystem health, which requires an understanding of how potential indicators respond across gradients of impairment. A linear response is preferable so that ecosystem health is simply proportional to the indicator measurement. However, linear responses to stressors are not expected for many ecosystem process measurements (Rama Rao, Singh & Mall 1979; Niyogi, Simon & Townsend 2003; Hagen, Webster & Benfield 2006) and are not necessary for distinguishing among sites with different states of ecosystem health. Nevertheless, the response does need to be predictable (Boulton 1999; Norris & Hawkins 2000). In this study we measured the responses of ecosystem metabolism, leaf decomposition, cotton strength loss, wood decomposition, and invertebrate community composition across a gradient of catchment impairment. We specifically attempted to answer two questions: (i) were there consistent responses to an impairment gradient among contrasting functional indicators of river ecosystem health, and (ii) what were the links between structural and functional indicators of river ecosystem health?

Methods

Study Sites

The study was conducted in two river catchments in New Zealand – the Motueka River in the upper South Island and the Mangaokewa Stream in the central North Island (Fig. 1). The Motueka River drains a catchment area of 2200 km^2 and flows in a northerly direction for 110 km from the headwaters to the sea. Mean annual rainfall ranges from <1 000 mm $year^{-1}$ on the eastern side of the catchment to 3 500 mm $year^{-1}$ on the western side. Median discharge at the bottom of the catchment is about 47 $m^3\ s^{-1}$ with a mean annual low flow of about 13 $m^3\ s^{-1}$. Land use is varied and includes native forest in the southern and western headwaters, plantation forest across much of the eastern and central part of the catchment, and pastoral farming and horticulture along the valley floors. The Motueka catchment is geologically complex with a mix of ultramafic and sedimentary rock in the southeastern headwaters, a complex array of sedimentary rocks underlying the western tributaries, a band of granitic rock down the western centre of the catchment, and a large band of alluvial gravel and clay down the eastern centre of the catchment (Young *et al.* 2005). The catchment is sparsely populated with a total population of around 12

000, most of whom live in the town of Motueka near the river mouth. Ten sites throughout the catchment were chosen to encompass a range of catchment modification and were distributed over 95 km of stream length (Fig. 1). Water quality is relatively good, but is closely related to the amount of agricultural development in the catchment (Table 1; Young *et al.* 2005).

The Mangaokewa Stream is a tributary of the Waikato River and drains an area of 65 km^2 in the North Island (Fig. 1). Mean annual rainfall surrounding the Mangaokewa Stream is about 1 450 mm $year^{-1}$. Intensive pastoral farming is the primary land use throughout the catchment and is the main stressor influencing Mangaokewa-1 (Table 1). Some discharges associated with urbanization affect Mangaokewa-2 and Mangaokewa-3, while treated sewage from the small town of Te Kuiti (population 4 400) is discharged into the river about 500 m upstream of Mangaokewa-4, and has been associated with elevated levels of dissolved reactive phosphorus and ammoniacal nitrogen (Table 1). The Mangaokewa Catchment is also geologically diverse with the headwaters draining pumice and tephra, while the middle/lower reaches pass through areas dominated by an array of sedimentary rock. The sites covered a stream length of 15 km and represented a gradient of cumulative stress comprising rural modification and urbanization with associated industrial inputs such that each site on the Mangaokewa had a distinctive stressor profile.

Catchment land use above each site was determined from the New Zealand Land Cover Database 2 (LCDB2, Terralink NZ Ltd). Land cover classes from the LCDB2 were condensed into %Native Forest, %Plantation Forest, %Pasture & Horticulture, and %Urban (Table 1). These data were used to calculate a weighted landuse stress score based on the ranked significance to stream health of different landuse impacts: urban (weighting factor = 3) > pasture/horticulture (2) > plantation forestry (no weighting) as described by Collier (2008). Scores over 200 indicate entirely developed catchments with a mix of urban and pastoral development, whereas scores < 100 indicate predominantly forested catchments. Water quality has been measured quarterly at six of the sites in the Motueka catchment (Wangapeka-1, Wangapeka-2, Motupiko, Motueka-1, Motueka-2, Motueka-3) since 2000 by the Tasman District Council as part of their regional water quality monitoring program (Young *et al.* 2005). Similarly, Environment Waikato has measured water quality monthly at Mangaokewa-2 since 2002, and sporadic data exist for other sites along this stream (Table 1).

A variety of functional and structural indicators of river ecosystem health were measured including ecosystem metabolism, organic matter decomposition (leaves, cotton and wood), and the composition of stream invertebrate communities. These indicators are either considered to have potential for stream health assessments (Boulton & Quinn 2000; Bunn & Davies 2000; Gessner & Chauvet 2002; Fellows *et al.* 2006; Paul *et al.* 2006; Young *et al.* 2008), or are already commonly used for stream health assessments (Barbour *et al.* 1999; Boothroyd & Stark 2000). Most indicators were measured at all sites, however, leaf decomposition was only measured at the Motueka sites. All measurements were made during the austral summer.

Ecosystem metabolism

Ecosystem metabolism, the combination of primary production and ecosystem respiration, was estimated using the single-station open-channel approach which requires measurement of the natural changes in dissolved oxygen concentration at the site over at least a 24-hour period (Owens 1974; Young & Huryn 1996). Many of the sites were too large to feasibly use the two-station open-system approach (Marzolf, Mulholland & Steinman 1994; Young & Huryn 1998). Oxygen concentration and temperature were recorded once every 10 minutes using a YSI 6920/6000 environmental monitoring system or a Hydrolab Datasonde 4. During measurements, a sonde was deployed at each site in a location as close as possible to the thalweg, and was chained to the bank or other suitable solid substrates. Prior to sampling, the sondes were calibrated in water-saturated air according to the manufacturer's instructions.

While recording oxygen concentrations, photosynthetically active radiation was measured every 15 seconds with a LI-COR quantum sensor and logged every 10 minutes using a LI-COR logger. This was done to determine the onset of darkness and daylight for calculation of ecosystem respiration and gross photosynthetic rate. Loggers were placed at locations considered representative of the light conditions prevailing upstream of the sites. An estimate of the average depth of each site was calculated using at least five measurements of depth at each of five cross-sections spaced at regular intervals upstream of the sonde to cover the local variation in channel morphology.

Metabolism values were calculated using a spreadsheet model as follows. Mean daily ecosystem respiration (ER) and the reaeration coefficient (*k*) were determined using the nighttime regression method (Owens 1974) which uses only data collected in the dark (< 2 µmol m^{-2} s^{-1}). The rate of change of oxygen concentration over short intervals was regressed against the oxygen deficit to yield:

$$dO/dt = \mathrm{ER} + kD$$

where dO/dt is the rate of change of oxygen concentration (g m^{-3} s^{-1}), ER is the ecosystem respiration rate (g m^{-3} s^{-1}), k is the reaeration coefficient (s^{-1}), and D is the oxygen deficit (g m^{-3}). The slope of the regression line estimates k while the y-intercept estimates ER (Kosinski 1984).

The reaeration coefficient and ecosystem respiration rate obtained were then used to determine gross photosynthetic rate over the sampling interval using:

$$\mathrm{GPP}_t = dO/dt + \mathrm{ER} - kD$$

where GPP_t is the gross photosynthetic rate (g m^{-3} s^{-1}) over time interval t. To compensate for daily temperature fluctuation, ER was assumed to double with a 10°C increase in temperature (Phinney & McIntire 1965) while the reaeration rate was assumed to increase by 2.41% per degree (Kilpatrick *et al.* 1989). Daily gross primary production (GPP, g m^{-3} day^{-1}) was estimated as the integral of all temperature-corrected photosynthetic rates during daylight (Wiley, Osbourne & Larimore 1990). This analysis gave values of production and respiration per unit volume. An areal estimate was obtained by multiplying the volume based estimates by average reach depth (m) which allowed comparison among sites with different depths. The P/R ratio was calculated as GPP/ER, while net ecosystem metabolism (NEM) was calculated as the difference between GPP and ER.

Organic matter decomposition

Three different organic substrates were deployed together in run/riffle habitat near the centre of the channel at each site to assess decomposition rates: leaves, strips of cotton cloth, and wooden sticks. Mahoe (*Melicytus ramiflorus* Forster) leaves were used to measure leaf decay because this tree species is relatively common throughout New Zealand and has been used in previous studies of leaf decomposition there (Linklater 1995; Parkyn & Winterbourn 1997; Hicks & Laboyrie 1999; Quinn *et al.* 2000). Mahoe leaf breakdown rates appear to be similar to those of fast-decaying leaf species (e.g. *Alnus glutinosa* L.) commonly used in Northern Hemisphere studies of leaf decomposition. The leaves were picked from a single tree to minimize variability among leaves and air-dried for 2 weeks before being transferred to 5 mm mesh bags. Each leaf pack contained 3-5 g of air-dried leaves and the weight of the contents of each bag was recorded to the nearest 0.001 g. Five replicate leaf packs were deployed at each site in the Motueka River for 1 month, but no leaf packs were deployed at any of the sites in the Mangaokewa Stream. Water temperature loggers were deployed at all sites so the effects of differences in water temperature could be compensated for in the analysis of organic matter decomposition by using degree-days as well as days as measures of exposure time (e.g. Minshall *et al.* 1983).

After retrieval, each leaf pack was placed in a separate plastic bag, stored on ice during transport and subsequently frozen until analysis. Any sediment, algae or invertebrates associated with the leaf material were gently washed from the leaves and discarded. The toughness of the leaves in each bag was determined using a penetrometer, which measures the weight required to force a blunt pin through a leaf (Young, Huryn & Townsend 1994). Five toughness measurements on different leaves were recorded for each leaf pack. Care was taken to ensure that the toughness measurements were not taken from parts of the leaf dominated by thick veins. After toughness measurements, the leaf material was dried to a constant weight in a 60°C forced-draft oven for at least 3 days. The dried leaf material was then weighed (to the nearest 0.001 g), ashed in a 550°C furnace and then reweighed to determine the ash-free dry mass (AFDM). To estimate the initial AFDM and toughness of leaves in each leaf pack we soaked five pre-weighed leaf packs for 24 hours in tap water to allow some initial leaching and then processed them in the same way as the other leaf packs. The post-leaching AFDM of these packs averaged 77% (range 76-80%) of their initial air-dry weight. This correction factor was then applied to all other leaf packs and accounted for the difference between air-dry weights and AFDM, plus the effects of initial leaching. Exponential decay coefficients for mass loss and toughness loss were determined using the equation presented in Petersen & Cummins (1974) using both days and degree-days as the time variables.

The cotton strips that we used in the Motueka River were standard Shirley Soil Burial Test fabric (Shirley Dyeing and Finishing Ltd, Hyde, U.K.), which is 100% combed cotton and has a series of coloured threads incorporated into the weave of the material so that strips can be frayed to a standard width (100 threads). Unfortunately, this material is no longer available, so we used some similar material with the same initial tensile strength in the Mangaokewa Stream. Five replicate cotton strips (4 cm wide x 10 cm long) were tethered at one end to anchor points (usually metal stakes) in run/riffle habitat at each site for 7 days. Following removal, they were stored on ice during transport and then subsequently frozen until analysis. After thawing the cotton strips were gently washed and dried at 20°C for 24 hours in a forced-draft oven. Threads were frayed from each side of the strips leaving a width of exactly 3 cm (100 threads). The tensile strength of each strip was measured on a tensometer. The initial tensile strength of the strips was determined using a set of control strips that were soaked in tap water for one day, and then frozen and processed in the same way as the other strips. The loss of tensile strength was reported in terms of exponential decay coefficients in the same way as the leaf breakdown data.

The wooden sticks that we used were birch wood (*Betula platyphylla* Sukaczev) coffee stirrer sticks (114 x 10 x 2 mm). Each stick was labeled with a permanent marker pen and then a hole was drilled at one end of the stick. The air-dried mass of each stick was measured (to the nearest 0.0001 g) and then five sticks were tied together, along with a plastic label, using nylon string. Short lengths (1 cm) of drinking straws were used to keep the individual sticks separated in the groups of five sticks. We deployed three to five groups of five sticks at each site for 3 months on the same stake as the cotton strips. Each group of sticks was weighed down to keep the sticks submerged on the river bed. Following retrieval, the sticks were kept on ice and then frozen until analysis. After thawing the sticks were gently washed and then dried to constant weight in a 60°C forced-draft oven and re-weighed. A set of control sticks was oven-dried to determine the difference between air-dry weight and oven-dry weight, which averaged 90% (range 89-90%). This correction factor was used to estimate initial oven-dry weights for the sticks that were deployed. Decay rates were reported as exponential decay coefficients in the same way as the leaf breakdown data.

Invertebrate community composition

The composition of the invertebrate community at each site was determined from single samples collected with a 0.5 mm mesh D-frame net using standard New Zealand invertebrate sampling protocols (Stark *et al.* 2001). This level of sampling intensity is typical of many biomonitoring programs where the focus is on community composition, rather than invertebrate densities (Boothroyd & Stark 2000). Samples were preserved in 70% alcohol and then invertebrates were identified to the lowest possible taxonomic level, counted and recorded. Three metrics were used to describe the invertebrate community composition: %EPT (Ephemeroptera, Plecoptera, and Trichoptera excluding Hydroptilidae) taxa in the samples, a biotic index that combines the presence/absence of particular taxa scored by their sensitivity to organic pollution (MCI, Stark 1985), and a semi-quantitative version of the same biotic index that gives weightings to abundance classes (SQMCI, Stark 1998).

Data analysis

Relationships between the landuse stress score and the measured indicators were assessed using linear and quadratic functions. Residuals were examined to ensure that outliers were not responsible for driving the relationships. A partial F-Statistic was used to determine if adding a second-order term to the model significantly improved the fit to the observed data compared with a linear model (Quinn & Keough 2002).

Redundancy analysis was also used to explore the relationships among the different indicators of river ecosystem health and between variables describing the impairment gradient (Zuur, Ieno & Smith 2007). Because leaf mass loss was not measured at any of the Mangaokewa sites, leaf decay rates were not included in this analysis. Similarly, because different cotton cloth was used in the Motueka and Mangaokewa rivers, cotton tensile strength loss rates were not included in this analysis. In situations where several indicators were calculated using the same data only one indicator was chosen for this analysis. For example, only the temperature-corrected wood mass loss rates were included in the analysis. Therefore indicators chosen for this analysis were GPP, ER, P/R, NEM, temperature-corrected wood mass loss, and %EPT taxa. Variables describing the gradient of impairment across the sites included: the landuse stress score, dissolved inorganic nitrogen (DIN), dissolved reactive phosphorus (DRP), and water clarity. There was a considerable difference in the size of catchments above each site, therefore we also included catchment area to account for this variability. Before analysis, %EPT taxa was arcsine√x transformed, whereas the landuse stress score, DIN and DRP were $\log_e$ transformed to improve normality.

Results

Ecosystem metabolism

Rates of GPP and ER ranged from 0.1-7.0 gO_2 m^{-2} day^{-1} and 0.34-16.5 gO_2 m^{-2} day^{-1}, respectively (Fig. 2). Rates of GPP were variable across the impairment gradient with relatively high rates in areas of low landuse stress, sites with some modification and also at the most impaired sites (Fig. 2). Linear and quadratic relationships between the landuse stress score and GPP were not significant ($P > 0.05$). In contrast, rates of ER increased across the impairment gradient (Fig. 2). The partial F-statistic comparing linear and quadratic relationships indicated that the quadratic model was no better than the linear relationship for explaining variation in ER.

Ratios of P/R ranged from 0.1 to 1.5, while rates of NEM ranged from -12.6-2.0 gO_2 m^{-2} day^{-1}. Both of these metrics indicated that sites at the low and high ends of the impairment gradient were heterotrophic with respiration rates presumably relying on organic matter sources from upstream or the surrounding catchment. In contrast, sites in the middle of the impairment gradient were more indicative of autotrophic conditions with respiration potentially supported by autochthonous production (Fig. 2). In both cases the partial F-statistic indicated that second-order relationships provided a significantly better fit to the observed data than linear models ($P < 0.05$).

Organic matter decomposition

Mass loss rates of mahoe leaves ranged from 0.04-0.07 day^{-1} and 0.002-0.005 degree-day^{-1}. As leaf decomposition data were available only from the sites in the Motueka catchment, it was difficult to judge a response across the entire impairment gradient. In the absence of temperature compensation, no significant relationships were evident between the landuse stress score and leaf mass loss or leaf toughness loss when calculated per day (Fig. 3). However, a linear relationship between landuse stress and leaf mass loss per degree-day was evident (Fig. 3), suggesting that any effect of the impairment gradient on leaf mass loss among the Motueka sites was masked by water temperature. The partial *F*-statistic comparing linear and quadratic relationships indicated that the quadratic model was no better than the linear relationship for explaining variation in leaf mass loss per degree-day.

The decline in tensile strength of the cotton strips ranged from 0.02-0.40 day^{-1} and 0.002-0.020 degree-day^{-1} (Fig. 4). Tensile strength loss rates were substantially higher at the Mangaokewa sites than the Motueka sites, perhaps suggesting a difference in decay rates between the Shirley Soil Burial Test fabric used at the Motueka sites and the alternative material used in the Mangaokewa. Therefore, data from the two catchments was considered separately. There was no significant relationship between landuse stress and tensile strength loss for the Motueka sites, but a significant increase in tensile strength loss with landuse stress within the Mangaokewa sites (Fig. 4). Decomposition was so fast at the two most impaired sites in the Mangaokewa Stream that the tensile strength of all the cotton strips was below the detection level of the tensometer after 7 days deployment.

The rate of mass loss for wooden sticks was equivalent to zero at one site (Motueka-3), which was substantially different from any of the other sites and therefore removed from further analysis as it was assumed to reflect a period of exposure during low river flows. Mass loss rates of wooden sticks ranged from 0.001-0.008 day^{-1} and 0.00007-0.00037 $degree\text{-}day^{-1}$ (Fig. 5). Stick mass loss per day increased across the impairment gradient, whereas temperature-compensated mass loss rates per degree-day showed a U-shaped response with lowest decay rates at intermediate levels of landuse stress (Fig. 5). The partial F-statistic comparing linear and quadratic relationships indicated that the quadratic model was no better than the linear relationship for explaining variation in stick mass loss per day, however there was an indication that the quadratic relationship provided a better fit than the linear relationship for the data on stick mass loss per degree-day ($P = 0.06$).

Invertebrate community composition

All three metrics describing the composition of the invertebrate community showed a similar response to the impairment gradient (Fig. 6). Sensitive high-scoring species were more common at sites with low landuse stress score, whereas pollution-tolerant, low-scoring species were more common at impaired sites. MCI and SQMCI scores greater than 120 and 6, respectively, are considered to reflect clean water, whereas MCI and SQMCI scores less than 100 and 5, respectively, indicate probable moderate or severe pollution (Boothroyd & Stark 2000). Scores between these extremes are considered to represent mild impairment. Therefore the sites used in this study ranged from excellent to poor water quality. Partial F-statistics comparing linear and quadratic relationships indicated that quadratic models were no better than the linear relationships for explaining variation in any of the invertebrate community metrics.

Relationships among indicators and environmental variables

The first axis of a redundancy analysis explained 50.7% of the variation in the data (Fig. 7). Axis 1 showed a positive correlation with landuse stress score, DIN concentration, DRP concentration, and a negative correlation with water clarity (Fig. 7). Catchment location covaried with landuse modification along axis 1. The second axis explained 18.5% of the variation in the data and was associated with catchment area and DIN concentration, although not significantly correlated. For the river health indicators, wooden stick mass loss, %EPT taxa, P/R and NEM were correlated with the first axis scores, whereas ER and GPP were correlated with the second axis suggesting that GPP, in particular, was responding to differences in stream size among the sites (Fig. 7). The redundancy analysis indicated that DRP concentration, catchment area and landuse stress score were the most important variables explaining the observed variation in the river health indicators.

Discussion

Factors affecting indicator response

Catchment location in this study co-varied with impairment, such that all highly impacted sites occurred in one catchment (Mangaokewa). Therefore, we cannot rule out a potential effect of geographic variation of the observed responses to the landuse stress gradient. However, we are confident that stressors associated with land use were the principal drivers of the relationships observed. This is supported for invertebrates by the findings of Quinn & Hickey (1990) who reported that land use was the key variable driving community composition in a nationwide survey of New Zealand streams, over-riding geographic patterns between islands or ecoregions. Similarly, Young *et al.* (2005) considered the effects of land use and geology on water quality in the Motueka catchment and found that land use was the primary determinant of water quality variables that are likely to influence river health (e.g., nutrient concentrations, turbidity and thermal regime), whereas geology was primarily responsible for less influential variables such as conductivity and pH. Nevertheless spatial factors, including geology and climate, may account for some of the variability in the response of the different indicators across the impairment gradient.

Other factors also have the potential to alter indicator responses to the severity of anthropogenic stressors. For example, ecosystem metabolism can vary seasonally or in response to bed-moving spates (Uehlinger 2006). The results obtained in the present study pertain to the austral summer which is pertinent to documenting landuse impacts in New Zealand because of the increased likelihood of sustained low flows, higher water temperatures and increased light levels leading to proliferations of plant growth. The variation in stream size among sites in this study is likely to have played an important role in affecting rates of GPP (Fig. 7), reflecting the well-documented effects of an increased channel size which reduces the shading from riparian vegetation (Vannote *et al.* 1980; Naiman 1983; Bott *et al.* 1985; Naiman *et al.* 1987; Minshall *et al.* 1992; McTammany *et al.* 2003).

Linear and non-linear responses

There are a variety of reasons why non-linear responses might be expected across impairment gradients. In the case of leaf and wood breakdown, the mechanism of decay is likely to vary substantially depending on the position of the site along the impairment gradient (Paul *et al.* 2007). In healthy systems, leaf decay may be primarily mediated by the activities of shredding invertebrates (Sponseller & Benfield 2001), whereas at more enriched sites decay would be expected to be accelerated by increased microbial activity (Meyer & Johnson 1983; Young *et al.* 2004; Suberkropp & Chauvet 1995; Gratton & Suberkropp 2001; Pascoal *et al.* 2003, Bergfur *et al.* 2007). Although leaf mass loss was not measured at the Mangaokewa sites, measurements at other sites in the Waikato region showed a positive correlation over the range 60-90% pasture in the catchment upstream (range of k values = 0.05-0.09 day^{-1} to 0.0025-0.0040 degree-day^{-1}; authors' unpublished data), suggesting that the response trajectory observed at the most impaired Motueka sites (see Fig. 3) would likely have continued at higher levels of landuse stress to give a U-shaped response across the impairment gradient. High numbers of potential shredding invertebrates were found in some of the sites with low levels of impairment, but there was no significant correlation between landuse stress score and shredder abundance in the Motueka streams (R. Young, unpublished data). High densities of snails, which can rasp leaf surfaces and contribute to their breakdown (Collier & Winterbourn 1986), are thought to be responsible for the high leaf mass loss rates observed at other sites in the Waikato region.

The shape of the response to impairment will also depend on the nature of the stressors involved. The non-linear response of P/R ratios, NEM and to a lesser extent GPP that we observed may be related to nutrients initially stimulating rates of GPP, whereas at higher levels of impairment the effects of decreased water clarity and sediment deposition may have been suppressing production as has been observed elsewhere (Wiley *et al.* 1990; Davies-Colley *et al.* 1992; Young & Huryn 1996), although as noted earlier GPP was apparently also related to stream size. Similar stressor responses have also been observed for organic matter decomposition, with a stimulation of decay rates at intermediate levels of impairment and then a decline in decay rates when high sediment inputs bury organic matter (Niyogi *et al.* 2003; Hagen *et al.* 2006). Invertebrate communities also appear to respond positively initially to nutrient stimulation, but negatively to increased fine sediment leading to non-linear responses to some stress gradients (Niyogi *et al.* 2007). Documentation of a broader array of habitat variables may have helped elucidate causal factors behind the observed relationships, underscoring the importance of incorporating habitat assessments with functional measurements of ecosystem health.

Implications for using measurements of ecosystem function for stream health monitoring

Our results suggest that (i) non-linear responses to catchment impairment are likely and need to be considered when interpreting measurements of ecosystem function, and (ii) contrasting indicators can respond in quite different ways to catchment impairment. Effective interpretation of measurements requires a good understanding of the likely stressors involved, and also an understanding of the likely mechanisms underlying observed responses to the stressor(s).

For the impairment gradient identified in this study, it appeared that ER, stick mass loss per day, leaf mass loss per degree-day, and indices based on invertebrate community composition varied in a linear fashion. Assuming this type of response is typical for a range of different impairment gradients, then the interpretation of the measurements is relatively easy with ecosystem health directly proportional to indicator measurements. A reference site approach can be used to compare with the results from test sites, or alternatively criteria that may be applied more widely could be developed and used as has been done with invertebrate community composition metrics (Maxted *et al.* 2000; Gessner & Chauvet 2002; Young *et al.* 2008). As we have shown, the response of the invertebrate metrics is often linear across impairment gradients, which perhaps explains the wide adoption of these metrics for river health assessment. However, some invertebrate metrics can be highly variable depending of the type and intensity of the stress leading to “wedge-shaped” or sinusoidal relationships, reflecting variations in the stability and/or persistence of communities under prevailing environmental conditions (Niyogi *et al.* 2007; Collier 2008).

For indicators that showed a quadratic response to the impairment gradient the interpretation of measurements is more difficult. In these situations indicator measurements at the low and high ends of the impairment gradient were often roughly equivalent, while measurements at intermediate levels of impairment were substantially different. This could be seen as an impediment to indicator interpretation (Hagen *et al.* 2006), but in most cases differences between the extremes of impaired and unimpaired are obvious and/or easy to detect using other indicators. In these situations the most valuable application of using such functional indicators would be to detect relatively subtle changes where the slope of the response curve is steep. For example, in the range from no impairment to intermediate impairment, indicator response may be proportional to ecosystem health and most sensitive to changes in stress intensity. This section of the impairment gradient may also be one of the most important if early signs of degradation are detected allowing management responses to be initiated before the degradation is too advanced, or for detecting initial moves away from degraded states necessary to document the early success of restoration activities (Palmer *et al.* 2005).

It is possible that some of the indicators responded to a threshold in the landuse stress gradient with little change over part of the gradient then a steep response at higher levels of impairment (e.g., Figs. 2, 5 & 6). Similar threshold responses have been observed for invertebrate communities in relation to nutrient and land use gradients (Niyogi *et al.* 2007). This type of response might be expected in situations where a stressor moves past a threshold, for example increasing nutrient concentrations promoting a change in periphyton community composition and/or a proliferation of benthic algae that affects habitat quality for other organisms (Biggs 2000). Further studies with sites distributed more evenly across the stress gradient may help to determine if threshold responses are common.

Another important consideration in river ecosystem health assessment is choosing the indicators that are most likely to respond to the stressor(s) that are potentially affecting the site of interest. Our results indicate that stick mass loss and potentially cotton strength loss appear to be sensitive to enrichment, whereas GPP may be less sensitive. On the other hand, GPP is likely to be more sensitive to removal of riparian vegetation if comparisons are restricted to sites of the same size, or if differences in stream size are compensated for in the analysis (Bunn *et al.* 1999; Fellows *et al.* 2006). Similarly, changes in bed morphology may have some impact on ecosystem metabolism or organic matter decomposition, but alternative indicators such as organic matter retention or sediment accumulation in leaf packs may be more effective indicators of the effects of habitat structure on ecosystem processes. Temperature compensation of decomposition rates may be appropriate in some situations where natural changes are expected across an impairment gradient. However, in other situations changes in the thermal regime may be the primary effect of the stressor and should be included in the analysis. Moreover, it may also be possible to compensate for degree of shading to determine whether riparian influences are the main regulator of GPP or whether other factors unrelated to shading are suppressing productivity.

Thus, the monitoring of functional indicators has the potential to provide an early warning on declining or improving conditions, and the possibility of linking observed changes to potential causal factors once relationships with environmental and habitat variables are more clearly understood. Although monitoring of functional indicators requires at least two visits to a site to deploy and retrieve instruments or substrates, and may require specialized equipment for measuring tensile strength, results are rapidly available as laboratory analyses are considerably less time-consuming than those typically required for macroinvertebrate sample processing. The added costs associated with increased field time may be more than compensated for by the increased information rapidly provided at critical stages of decline or recovery.

Relationships between structural and functional indices of river health

The structure and function of river ecosystems are intricately linked (Wallace *et al.* 1997; Covich *et al.* 2004), but responses to impairment can potentially vary with changes to structure only, function only, or both structure and function (Matthews *et al.*1982). Close linkages among structural and functional indices would be expected if there was a strong mechanistic link between these measures. For example, leaf litter decay is often affected by the abundance and diversity of shredding invertebrates (Wallace & Webster 1996), so a reasonable relationship might be expected between leaf decomposition rates and indices describing invertebrate community structure. However, common invertebrate-based indices do not necessarily focus on the presence/absence of particularly functional groups of invertebrates, reducing the likelihood of such linkages being detected. Furthermore, many ecosystem processes are primarily controlled by microbial activity rather than invertebrate community composition, and therefore any direct links between ecosystem process measures and structural indices based on invertebrate community composition would be relatively unlikely.

In the current study we found positive relationships between invertebrate-based indices and rates of NEM and P/R and negative relationships with ER, wood mass loss and leaf mass loss, but no relationships with the other functional indicators (Fig. 7). Relationships might be expected between structural indices and measurements of ecosystem processes if indices are derived from microbial community structure (Duarte *et al.* 2006) or are based on invertebrate functional traits (e.g., Dolédec *et al.* 2006). Our results emphasise that variability in response to impairment is likely among different ecosystem processes so consistent relationships among structure and functional indices of river ecosystem health should not be expected, and indeed may not be desirable if the aim is to minimize redundancy among ecological indicators.

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Table 1 Land use and water quality characteristics of the sampling sites. DRP = dissolved reactive phosphorus.

Site	Catchment Area (km^2)	%Native Forest	%Plantation Forest	%Pasture & Horticulture	%Urban	Landuse stress score	Water clarity (m)	DRP ($g\ m^{-3}$)	NO_3-N ($g\ m^{-3}$)	NH_4-N ($g\ m^{-3}$)
Rainy-1	55	96	2	2	0	6	7.5	0.010	0.015	0.009
Rainy-2	89	87	6	7	0	20	7.0	na	na	na
Motupiko	375	50	27	22	0	71	3.9	0.009	0.077	0.010
Motueka-1	939	50	31	17	0.1	65	5.7	0.007	0.225	0.008
Granity	26	100	0	0	0	0	na	na	na	na
Rolling	88	100	0	0	0	0	8.3	0.009	0.062	0.008
Wangapeka-1	309	99	0	0.5	0	1	7.8	0.005	0.043	0.010
Wangapeka-2	538	81	9	8	0	25	4.4	0.005	0.103	0.008
Motueka-2	1832	61	24	13	0.1	50	4.3	0.005	0.116	0.008
Motueka-3	2191	56	25	16	0.1	57	3.7	0.006	0.129	0.008
Mangaokewa-1	132	20	11	69	0	149	1.2	0.020	0.150	0.016
Mangaokewa-2	160	17	9	72	0	153	1.1	0.022	0.505	0.020
Mangaokewa-3	168	17	9	71	2	157	1.5	0.035	0.393	0.310
Mangaokewa-4	173	16	9	72	2	159	1.2	0.260	0.384	1.440
Mangaokewa-5	190	15	8	74	2	162	na	0.220	na	na

na, not available

Figure Legends

Fig. 1 The location of the sampling sites in the Motueka and Mangaokewa catchments.

Fig. 2 Response of gross primary production (GPP), ecosystem respiration (ER), the ratio of P/R and net ecosystem metabolism (NEM) across the landuse stress gradient.

Fig. 3 Response of mahoe leaf mass and toughness loss (±SE) across the landuse stress gradient. Decay rates are reported in terms of mass loss per day and temperature-compensated loss per degree-day.

Fig. 4 Response of cotton strip tensile strength loss (±SE) across the landuse stress gradient. Strength loss rates are reported in terms of mass loss per day and temperature-compensated loss per degree-day. Differences in the initial cotton material used in the Motueka and Mangaokewa sites meant that analysis was conducted separately for each catchment. Only the relationships in the Mangaokewa Catchment were significant.

Fig. 5 Response of wooden stick mass loss (±SE) across the landuse stress gradient. Decay rates are reported in terms of mass loss per day and temperature-compensated loss per degree-day. Data from one outlier was excluded from the analysis.

Fig. 6 Response of metrics describing invertebrate community composition across the gradient of impairment. The %EPT taxa metric refers to the percentage of taxa comprising mayflies, stoneflies and caddisflies (excluding Hydroptilidae). The MCI and

SQMCI are biotic indices commonly used in New Zealand that are calculated using taxon sensitivity scores for presence/absence or abundance class data, respectively.

Fig. 7 Redundancy analysis triplot of variables used to describe river ecosystem health and variables characterizing the landuse stress gradient. The location of sites in the ordination is also shown (exact position is at the centre of the site label). Bold lines/variables indicate physicochemical explanatory variables. DIN = dissolved inorganic nitrogen, DRP = dissolved reactive phosphorus, %EPT = percentage of macroinvertebrate taxa belonging to the Ephemeroptera, Plecoptera and Trichoptera (excluding Hydroptilidae), ER = ecosystem respiration, GPP = gross primary production, LUS = Landuse stress score, NEM = net ecosystem metabolism, P/R = production/respiration.

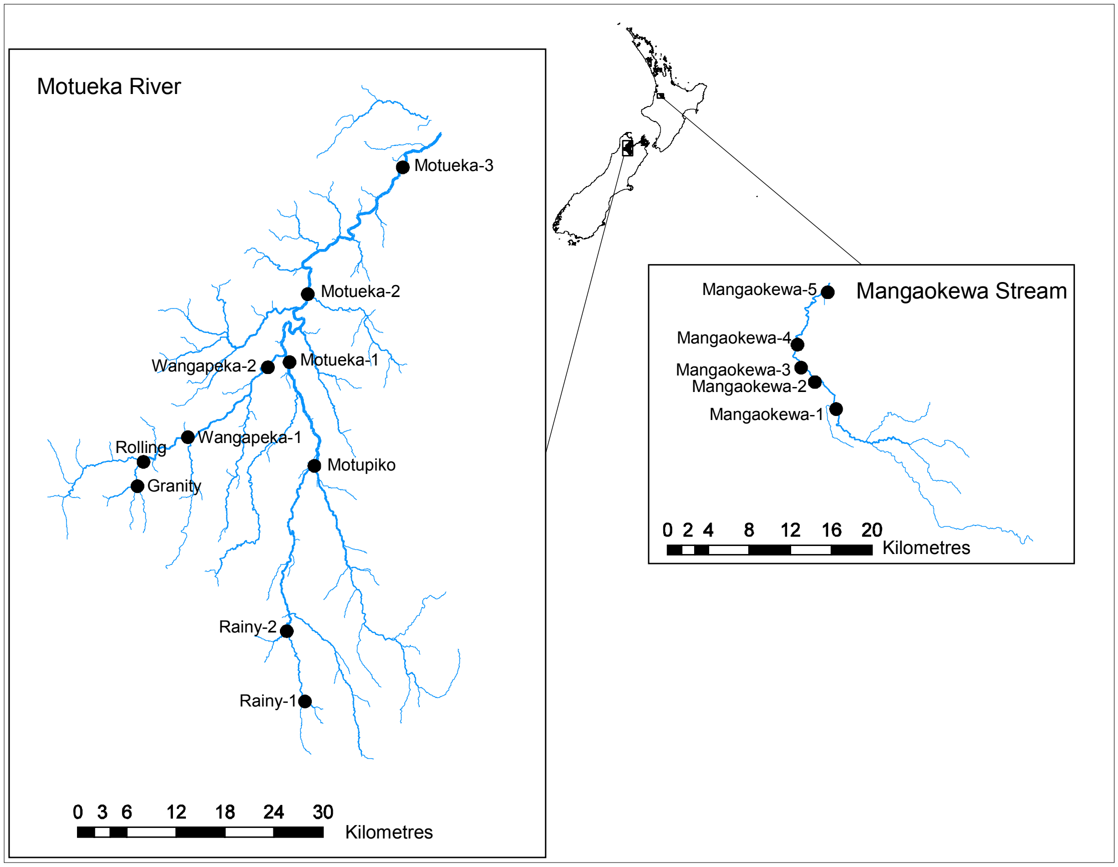

Motueka River
Motueka-3
Motueka-2
Wangapeka-2
Motueka-1
Rolling
Wangapeka-1
Granity
Motupiko
Rainy-2
Rainy-1
0 3 6 12 18 24 30
Kilometres
Mangaokewa Stream
Mangaokewa-5
Mangaokewa-4
Mangaokewa-3
Mangaokewa-2
Mangaokewa-1
0 2 4 8 12 16 20
Kilometres

Fig. 1

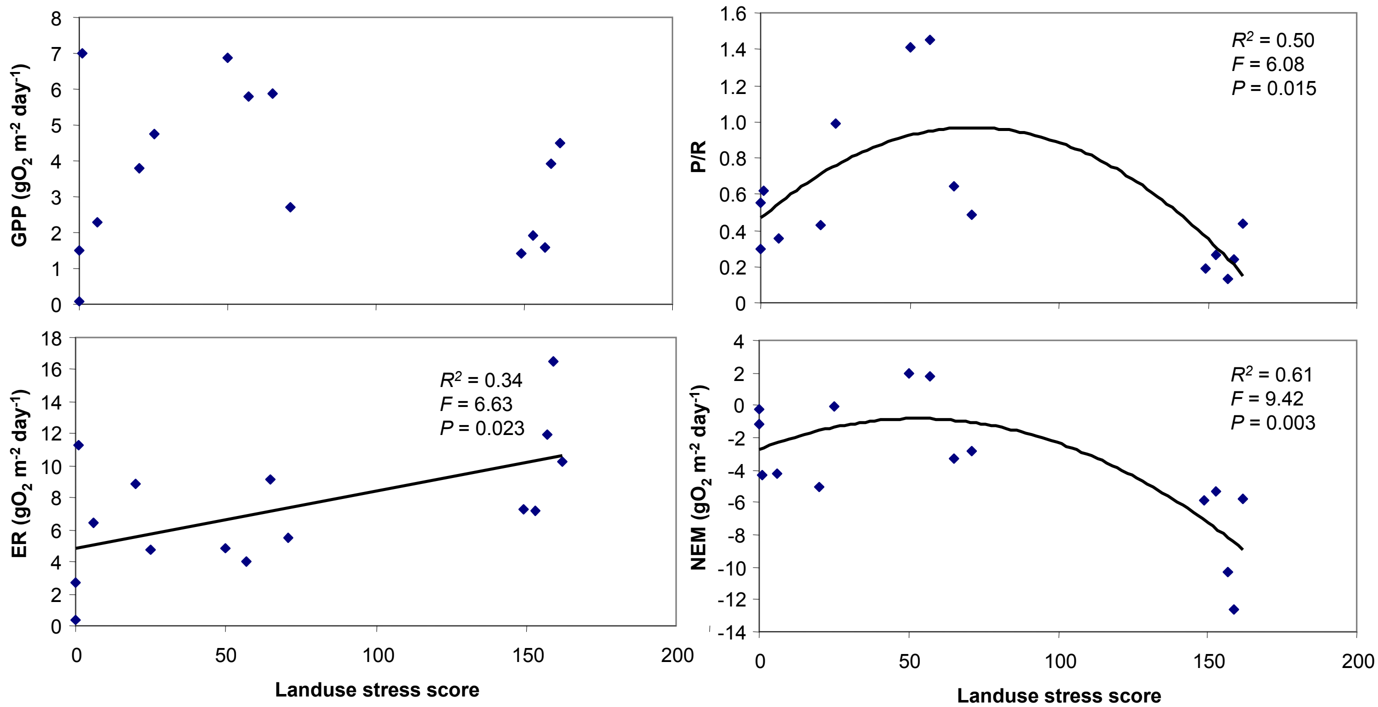

GPP (gO2 m-2 day-1)
P/R
R2 = 0.50
F = 6.08
P = 0.015
ER (gO2 m-2 day-1)
R2 = 0.34
F = 6.63
P = 0.023
NEM (gO2 m-2 day-1)
R2 = 0.61
F = 9.42
P = 0.003
Landuse stress score
Landuse stress score

Fig. 2

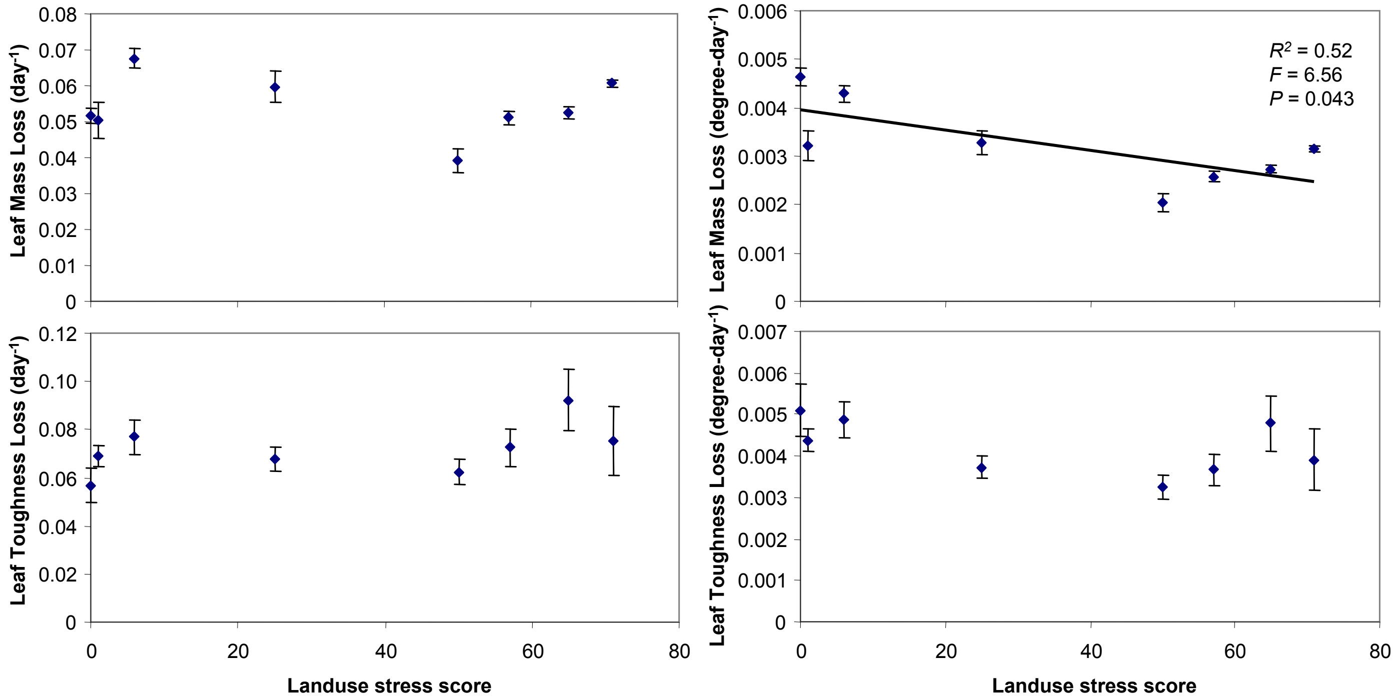
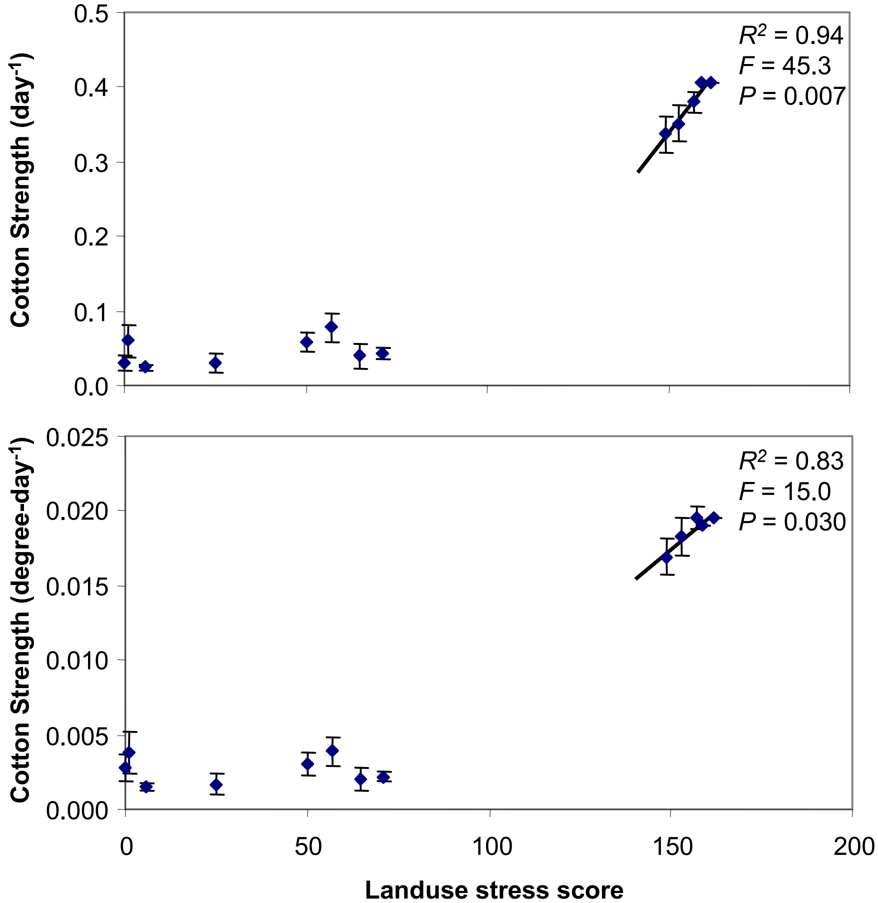

Leaf Mass Loss (day-1)
Leaf Mass Loss (degree-day-1)
R2 = 0.52
F = 6.56
P = 0.043
Leaf Toughness Loss (day-1)
Leaf Toughness Loss (degree-day-1)
Landuse stress score
Landuse stress score
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
0.006
0.005
0.004
0.003
0.002
0.001
0
0.12
0.10
0.08
0.06
0.04
0.02
0
0.007
0.006
0.005
0.004
0.003
0.002
0.001
0
0
20
40
60
80

Fig. 3



0.5
0.4
0.3
0.2
0.1
0.0
Cotton Strength (day-1)
R^2 = 0.94
F = 45.3
P = 0.007
0.025
0.020
0.015
0.010
0.005
0.000
Cotton Strength (degree-day-1)
R^2 = 0.83
F = 15.0
P = 0.030
0
50
100
150
200
Landuse stress score

Fig. 4

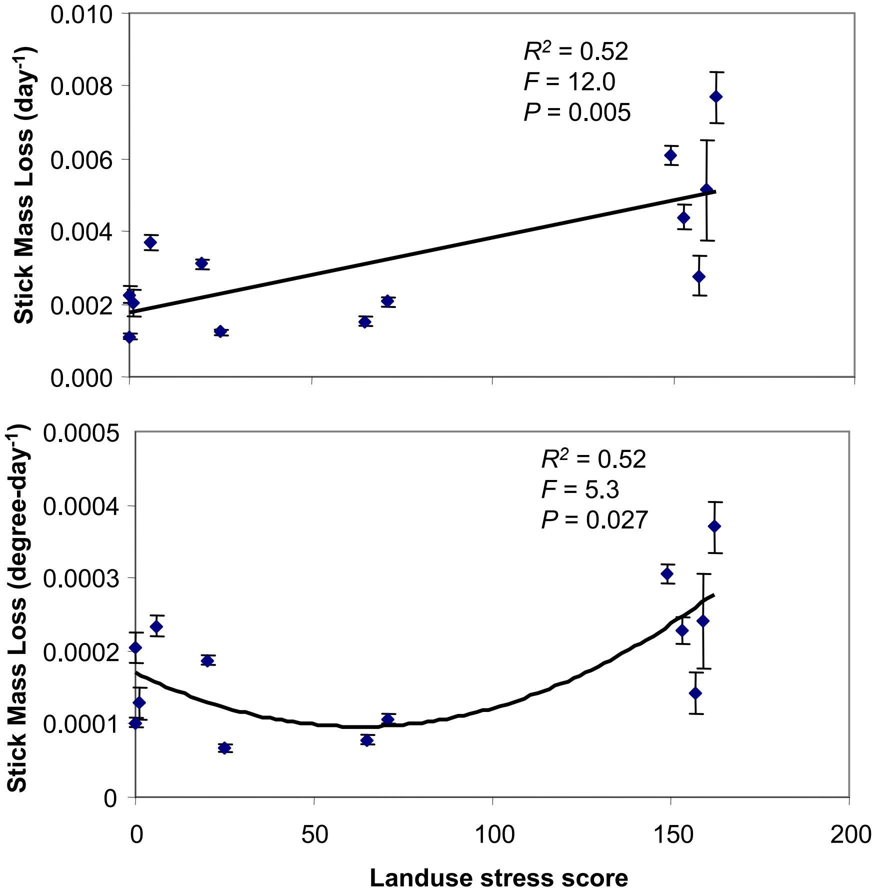

0.010
0.008
0.006
0.004
0.002
0.000
Stick Mass Loss (day-1)
R2 = 0.52
F = 12.0
P = 0.005
0.0005
0.0004
0.0003
0.0002
0.0001
0
Stick Mass Loss (degree-day-1)
R2 = 0.52
F = 5.3
P = 0.027
0
50
100
150
200
Landuse stress score

Fig. 5

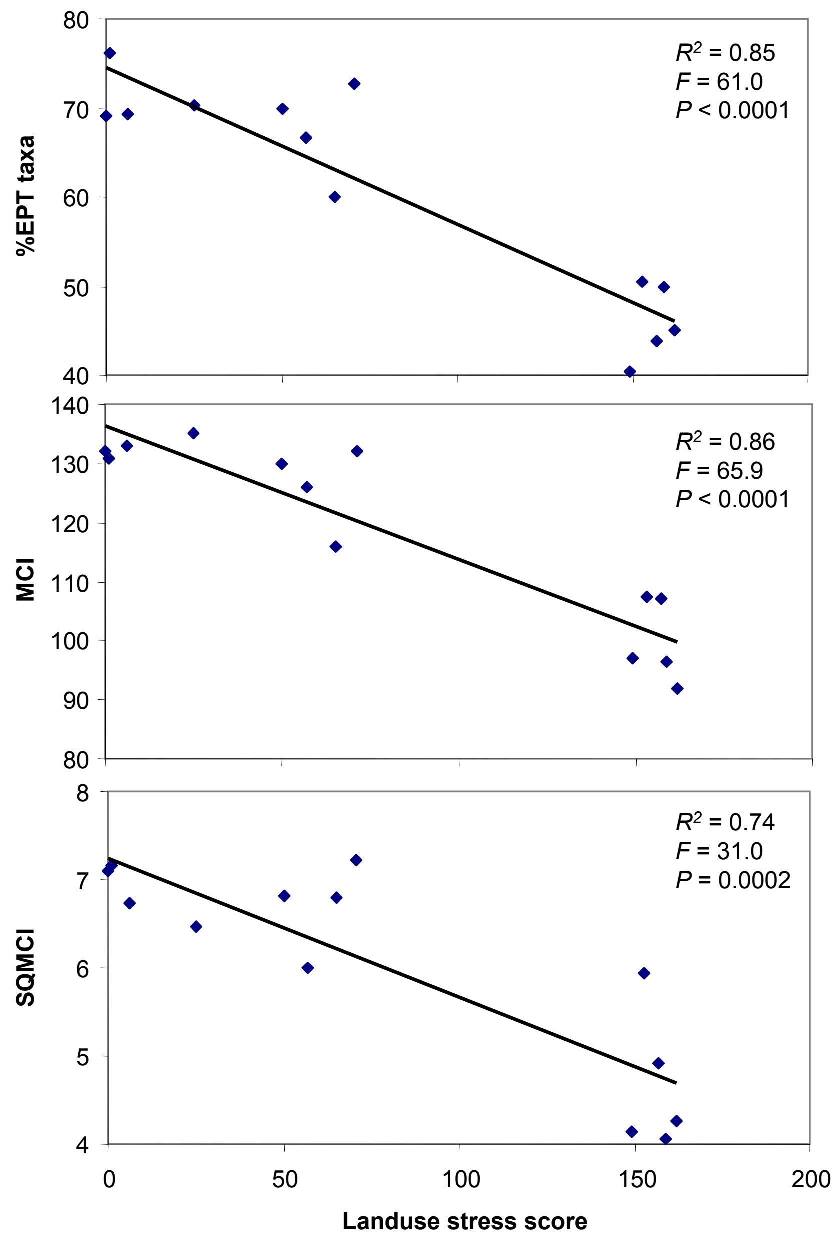
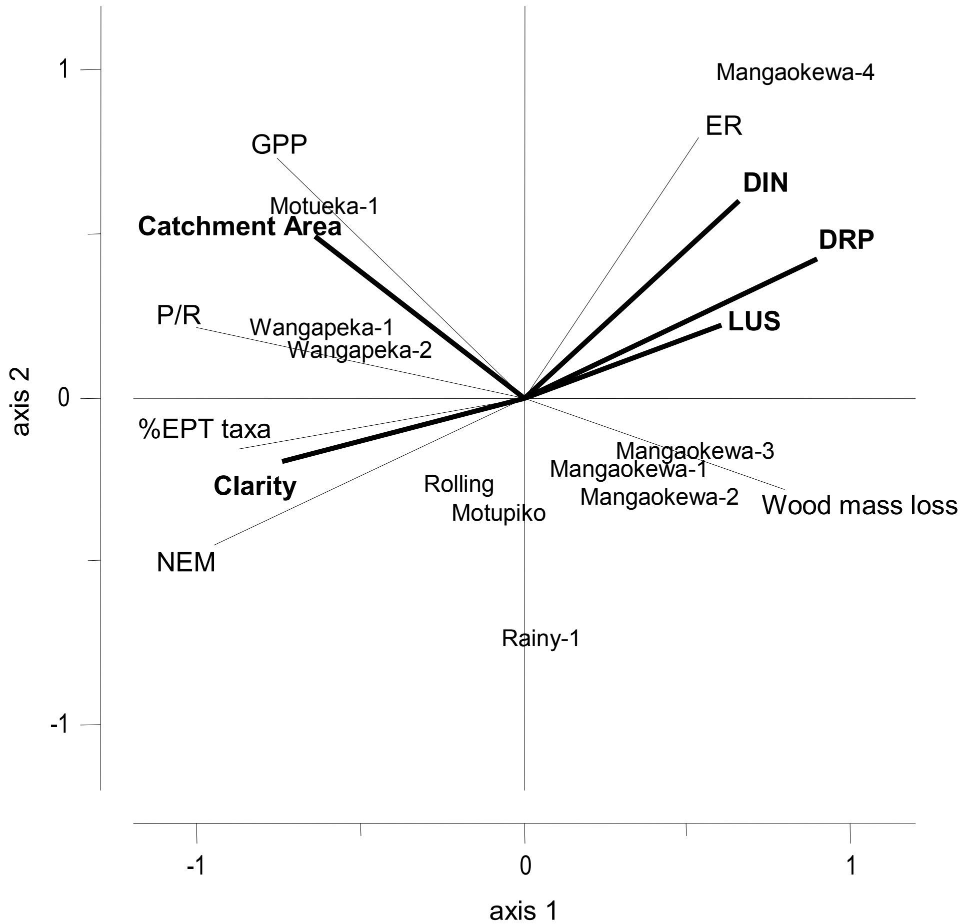

%EPT taxa
R^2 = 0.85
F = 61.0
P < 0.0001
MCI
R^2 = 0.86
F = 65.9
P < 0.0001
SQMCI
R^2 = 0.74
F = 31.0
P = 0.0002
Landuse stress score

Fig. 6



Mangaokewa-4
ER
DIN
DRP
LUS
GPP
Motueka-1
Catchment Area
P/R
Wangapeka-1
Wangapeka-2
%EPT taxa
Clarity
NEM
Rolling
Motupiko
Mangaokewa-3
Mangaokewa-1
Mangaokewa-2
Wood mass loss
Rainy-1
1
0
-1
axis 2
-1
0
1
axis 1

Fig. 7