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By Annika Wagenhoff, Christoph D. Matthaei and Colin R. Townsend,
Department of Zoology, University of Otago

Multiple-stressor effects on stream health in Southland streams and rivers

Introduction

Our understanding of the ecological health of agricultural streams and rivers, usually measured in terms of macroinvertebrate community structure (MCI, EPT richness), is far from complete. Given the ongoing intensification in the Southland region of agricultural land use (particularly dairying), the causes of poor ecological health of streams need to be fully understood if present impacts are to be mitigated and further degradation prevented.

Two major causes of poor ecological health of agricultural streams are increased nutrient and fine sediment inputs from surrounding land. For this reason we refer to nutrient concentrations in stream water and sediment cover on the bed as stressors. Impacted streams sometimes have high levels of deposited fine sediment but low to moderate nutrient concentrations, others have high nutrients but low to moderate sediment, and in some streams levels of both stressors are high. Fine sediment and nutrients can act as multiple stressors in complex, unpredictable ways such that, for example, nutrients may have little deleterious effect at low sediment levels but a strongly deleterious, synergistic effect when coupled with high sediment levels (Townsend et al. 2008). We define a cause-effect relationship as complex when multiple stressors interact with each other so that the outcome cannot be predicted by simply adding the individual effects of the single stressors.

Managers need to know about cause-effect relationships involving multiple stressors and ecological responses in order to improve environmental management by prioritising actions according to the likelihood and speed of achieving predicted positive outcomes. For example, management actions that reduce sediment inputs at a certain site might be more effective than actions that aim at lowering nutrient concentrations, or vice versa.

To investigate cause-effect relationships we need to quantify deposited fine sediment and nutrient concentrations. Regional councils typically monitor dissolved forms of the nutrients

nitrogen and phosphorus, although not always at stream health monitoring sites. However, fine sediment has rarely been recorded by regional councils to date. Environment Southland has not previously measured sediments in streams but is considering future routine measurement of fine sediments at stream health monitoring sites, with the further aims of tracking land-use intensification and its effects through time, determining baseline values for the development of sediment guidelines, and supporting technical guidelines for discharge plan rules relating to sediment input to waterways. The demand for annual monitoring at multiple sites highlights the need for both cost- and time-effective methods.

The aims of this report are (i) to present analyses of multiple-stressor effects of nutrients and deposited fine sediment on macroinvertebrate community indices commonly used as indicators of stream health by regional councils, in particular to determine whether cause-effect relationships are simple or complex and (ii) to compare different methods and measures of deposited fine sediment in terms of their ability to predict macroinvertebrate community indices but also their cost- and time-efficiency.

Methods

Field survey sites

Study sites were a subset of Environment Southland's long-term State-of-the-Environment (SoE) stream health monitoring programme in wadeable streams, consisting of 43 streams and rivers scattered around Southland. Stream order ranged from second to sixth order, wetted width varied between 2 and 45 m and sampled stream reach length varied from 5 to 56 m. Each site was visited once at baseflow conditions to take biological, water quality and deposited fine sediment samples from one specific riffle or run habitat. The survey was completed during a two-week period in the summer of 2008 (January).

According to New Zealand's River Environment Classification (REC), study sites fell into one of three geological categories: (1) hard sedimentary spatially dominant (15 sites), (2) alluvium spatially dominant (10 sites) or (3) soft sedimentary >25% (18 sites). The REC land cover categories for the sites were: (1) native forest spatially dominant (6 sites, 3 in each of geology categories 1 and 3), (2) tussock spatially dominant (2 sites, both in geology category 1) or (3) pasture cover >25% (35 sites: 10, 15 and 10 sites in geology categories 1, 2 and 3, respectively).

Biological sampling

At each site, a single semi-quantitative macroinvertebrate sample was taken with a D-shaped hand net (0.5 mm mesh size) as described in detail in Stark et al. (2001). In hard-bottomed streams, macroinvertebrates were collected with the foot-kicking method and sampling effort and area were standardised by disturbing bed substrate in 10 different locations of varying velocity regimes in the sampling reach and pooling the collected animals. In soft-bottomed streams, different habitat units such as bank margins, macrophytes and woody debris were sampled in proportion to their frequency of occurrence. Here the sampling method varied according to the habitat unit. From bank margins and macrophyte beds, animals were dislodged with the hand net and collected from the water column by subsequent sweeps of the hand net. Animals on woody debris were washed or picked off into the net. Samples were preserved in ethanol.

A fixed count of 200 macroinvertebrates per sample was identified in the laboratory to the lowest practicable taxonomic level for determination of community composition parameters and stream health indices. These included: (i) invertebrate taxon richness, (ii) EPT taxon richness (EPT = organisms in the orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)), (iii) MCI (Macroinvertebrate Community Index) and (iv) % EPT (relative abundance of individuals in the EPT orders).

Periphyton samples for community analysis were collected from areas with different current velocity regimes in each site by scraping biofilm from randomly selected stones in each area and preserving in Lugol's solution. Algae were identified in the laboratory to the lowest practicable taxonomic level for the determination of algal taxon richness.

Water quality

The nutrient status at each site was determined from a single water sample using standard methods for dissolved inorganic forms of nitrogen and phosphorus. These were the sum of NO_2-N, NO_3-N and NH_4-N, reported as dissolved inorganic nitrogen (DIN µg/L), and dissolved reactive phosphorus (DRP µg/L), respectively.

Deposited fine sediment

Three methods were used to quantify deposited fine sediment on the streambed at each study site: (1) % cover of fine sediment on the streambed visually estimated in the field, (2) mean depth of fine sediment covering the streambed and (3) suspendable inorganic sediment (SIS) as determined in the laboratory from samples taken by a Quorer (Quinn and Cooper 1997). Each of

the three measures was taken according to a similar sampling design. In brief, measures were taken at several locations along five equidistant transects distributed down the length of the site from which biological samples were taken. The values were then averaged to characterise the fine sediment status of the site. The sequence of sampling along each transect was as follows.

(1) % cover of fine sediment

The method for visual estimation of % cover of fine sediment (grain size < 2 mm) on the streambed using a viewing box is described in detail in the attached protocol. The % cover of fine sediment recorded for a site is the average of 10 estimates.

(2) Sediment depth

Sediment depth measurements (in mm) were taken at three random points (where fine sediment < 2 mm was present) within the same sampling quadrat for which sediment cover had been estimated. This was done by inserting a ruler into soft, permeable fine sediment without pressure until the underlying, coarser streambed substrate was reached. If fine sediment was less than 1 mm deep it was recorded as 0.5 mm. The sediment depth recorded for a site was the average of 30 measurements (3 at each of the 10 locations).

(3) Suspendable inorganic sediment

Samples of fine sediment that was re-suspendable by physical disturbance were collected from the uppermost layer of the bed substrate in five locations per site using the Quorer (Quinn and Cooper 1997) for subsequent quantitative analysis of suspendable inorganic sediment in the laboratory. In each location, a Quorer of 24 cm diameter was sealed tightly onto the streambed and 5 random water depth measurements were taken. Then substrate was disturbed to a depth of 5 cm with a screwdriver for 30 seconds and a 120-ml sub-sample of the slurry collected from within the enclosed water column. Two water samples were also taken to correct for background suspended solids. In the laboratory, fine sediment samples were dried, weighed, ashed at 550 °C and weighed again to determine the inorganic mass of the sediment as suspendable inorganic sediment per stream area sampled (SIS in g/m^2). The SIS value for each site represented the average of the 5 sampling locations. For more information about the Quorer method see http://www.niwascience.co.nz/ncwr/tools/quorer.

Statistical analysis

To explore the relationships between stream health indices and the two stressors (fine sediment and nutrients), we used simple and multiple linear regression analyses to calculate several models for each of the biological response variables (MCI, invertebrate taxon richness, EPT

richness, % EPT and algal taxon richness). The set of models was chosen to investigate and compare the fit of single-stressor models (that included either sediment or nutrients) with the fit of multiple-stressor models (that included both sediment and nutrients). To fully investigate the complexity of multiple-stressor models, an interaction term (fine sediment × nutrients) was also included. The set of models for each biological response variable was therefore as follows:

(1) $Y = b_0 + b_1 \times S + b_2 \times N + b_3 \times S \times N$

(2) $Y = b_0 + b_1 \times S + b_2 \times N$

(3) $Y = b_0 + b_1 \times S + b_2 \times S \times N$

(4) $Y = b_0 + b_1 \times N + b_2 \times S \times N$

(5) $Y = b_0 + b_1 \times S$

(6) $Y = b_0 + b_1 \times N$

(7) $Y = b_0 + b_1 \times S \times N$

In these models, Y is a biological response variable (MCI, invertebrate taxon richness, EPT richness, % EPT or algal taxon richness), S, N and S×N are the predictor variables (fine sediment, nutrients and the interaction between them), and the constants b_0, b_1, b_2 and b_3 are the regression parameters. Model forms 1, 2, 3, 4 and 7 are multiple-stressor models, while forms 5 and 6 are single-stressor models.

We were also interested in comparing the three measures of fine sediment (% sediment cover, sediment depth and SIS). Consequently, the set of seven models was run with each sediment measure. Furthermore, we wished to know whether nitrogen or phosphorus or both in combination were important drivers of our biological response variables. Thus, we also ran the seven models each with DIN, DRP and, finally, a nutrient index, $\log_{10}(\text{DIN})+\log_{10}(\text{DRP})$, as predictors. (This nutrient index is a variable that has been shown to be useful in explaining variation in macroinvertebrate community indices in previous research (Niyogi et al. 2007, Townsend et al. 2008).

Given the three sediment variables and three nutrient variables, we ran 51 models in total (3×3×5 models that contained both sediment and nutrients plus 3 sediment and 3 nutrient single-stressor models) for each of the response variables. A model was rejected if (1) it was not significantly different from the null model, namely $Y = b_0$ (tested with the F-statistic) or (2) if its regression parameter estimates were not significant (tested with the t-statistic). All significant models were retained and ranked according to Akaike's Information Criterion (AIC; Crawley

(2007)) to compare model performance and select the best overall model for each biological response variable.

The data distributions of all predictor variables were skewed to the right and so data were transformed to improve model performance. The transformed predictor variables were: (% sediment cover)$^{1/3}$, ln(sediment depth×10+1), ln(SIS), ln(DIN), ln(DRP) and $\log_{10}$(DIN)+$\log_{10}$(DRP). For the same reason, the response variable % EPT was transformed to arcsin(% EPT/100). All analyses were computed with statistical computer programme R (R Development Core Team 2008).

Results

Summary statistics for the 43 study sites

Summary statistics for nutrient concentrations and deposited fine sediment measures as well as biological response variables are presented in Table 1. All individual site scores for these variables are provided in Appendix 1.

Table 1: Summary statistics for nutrient concentrations, deposited fine sediment measures and macroinvertebrate and algal community indices (DIN=dissolved inorganic nitrogen, DRP=dissolved reactive phosphorus, SIS=suspendable inorganic sediment, MCI=Macroinvertebrate Community Index, EPT=Ephemeroptera, Plecoptera, Trichoptera).

Data	Minimum	1st Quantile	Median	Mean	3rd Quantile	Maximum
DIN (μg/L)	10	47	106	327	423	1928
DRP (μg/L)	3	7	9	24	25	250
% cover of fine sediment	0	1	6	14	16	100
Sediment depth (mm)	0.0	0.6	1.4	5.2	4.1	79.3
SIS (g/m^2)	23	163	361	827	765	10870
MCI	64	86	100	101	117	130
Invertebrate richness	8	15	18	19	22	29
EPT richness	2	6	9	9	11	19
% EPT	3	20	48	46	66	89
Algal taxon richness	2	10	13	13	16	25

Models for macroinvertebrate community indices

Selection of best models

From a longer list of statistically significant models (not shown), the best overall model for each biological response variable was selected for its lowest AIC value. This procedure was successful for MCI, EPT taxon richness and % EPT. None of the models for total invertebrate taxon richness was significant so this variable will not be considered further. The best models for MCI and the two EPT response variables, presented in Table 2, were of the same form ($Y = b_0 + b_1 \times N + b_2 \times S \times N$), each including two predictor variables, namely nutrients and the interaction term (the product of nutrients and fine sediment).

All three invertebrate response variables showed a positive relationship with nutrients and a negative relationship with the interaction term. The sediment variable that entered all three best models was SIS. The nutrient variable for MCI and EPT richness was the nutrient index $\log_{10}(DIN)+\log_{10}(DRP)$, whereas for % EPT it was DIN.

R-squared is the fraction of total variation in the data explained by the model or, in other words, the explanatory power of the model. The model for % EPT explained the greatest amount of variation of all response parameters, with an R-squared of 0.52 (52 % of the total variation in the % EPT data accounted for).

Table 2: Variables and parameters of best overall models of the form $Y = b_0 + b_1 \times N + b_2 \times S \times N$ (Y = response variable, N = nutrient predictor variable, S = sediment predictor variable, b_0 to b_3 = regression parameters, R^2 = R-squared value)

Y	N	S	b_0	b_1	b_2	R^2
MCI	$\log_{10}(DIN)+\log_{10}(DRP)$	ln(SIS)	120.14	10.73	– 2.80	0.46
EPT richness	$\log_{10}(DIN)+\log_{10}(DRP)$	ln(SIS)	11.79	2.34	– 0.53	0.34
% EPT	ln(DIN)	ln(SIS)	0.91	0.11	– 0.03	0.52

3-D graphical presentations of best models

To visualise the main and interactive effects of the two stressors on ecological response variables, the individual data points are presented in 3-D graphs (Figure 1.a-c). In these graphs, the best predictive models are plotted as regression surfaces, with the response variable (calculated from the model) plotted on the y-axis against the two predictor variables fine

sediment and nutrients on the x-axis and z-axis, respectively. If, for the sake of argument, nutrient concentration was the only predictor variable in a model, then the regression surface would be seen as a plane declining as nutrients increased but lying parallel to the sediment axis (because a change in fine sediment does not affect the response). If both nutrients and fine sediment were in a model, but not the interaction term, then the regression surface would also be a plane, this time declining with increasing sediment and declining with increasing nutrients. When a model includes a significant interaction term, the regression surface becomes distorted away from a simple plane.

The three figures representing our best models (Figure 1.a-c) look similar because each is based on the same form of model, including nutrients and the interaction term as predictor variables. In each case the biological response variable declines with increasing nutrient concentration, but the rate of decline is faster at higher sediment levels. It is also apparent that the adverse effect of sediment is greater at higher nutrient concentrations. In other words, there is no simple additive effect of the two stressors, but rather they interact in their influence on the response variable. The distortion away from a simple plane is a consequence of this interaction.

Multiple- vs. single-stressor models

For the response variables MCI, EPT richness and % EPT, there were respectively 22, 15 and 24 models that were statistically significant (out of a possible 51 in each case). We ranked these significant models according to their performance using AIC. Multiple-stressor models (model forms 1, 2, 3, 4 and 7, see Methods) were ranked highest. For MCI and EPT richness, the best single-stressor model was only ranked 9th (out of 22 and 15, respectively) while for % EPT the best single-stressor model was only ranked 15th (out of 24). For all three response variables, sediment single-stressor models ranked higher than nutrient single-stressor models, regardless of which measure of fine sediment was in the model. For EPT richness, none of the nutrient single-stressor models was significant.

The best overall model for % EPT explained 52 % of the variation compared to the best single-stressor model that only explained 37 % of the variation in the data. For MCI 48 % of the variation could be explained by the best overall model in comparison to 37 % by the best single-stressor model. For EPT richness 34 % was explained by the best overall model compared to 25 % by the best single-stressor model.

Simple or complex interactions in multiple-stressor models

None of the models with three terms (model form 1) was significant. However, multiple-stressor models that incorporated the complex interaction term and were statistically significant (model forms 3, 4 and 7) generally ranked more highly than models where the effects of sediment and nutrients were simply additive. In fact, the eight most highly ranked models for % EPT were those with the interaction term. For MCI and EPT richness, respectively, the four and two most highly ranked models included the interaction term. Models with simply additive effects of sediment and nutrients (model form 2) took ranks 5, 10 and 13 amongst MCI models and ranks 3, 5 and 10 amongst models for % EPT. None of the simple additive models was significant for EPT richness.

Comparing models with different fine sediment measures

The sediment predictor variable in the three best models was SIS. In fact, for % EPT the five most highly ranked models included SIS, while the two most highly ranked models for EPT richness included SIS. The best model including sediment cover was ranked 3 for EPT richness, 6 for % EPT and 8 for MCI. The best model including sediment depth was ranked 2 for MCI, 5 for EPT richness and 7 for % EPT.

In spite of the lower rankings of models with sediment cover or depth as the sediment predictor the explanatory power of these models was still high. For MCI the most highly ranked model with either sediment depth or cover as the sediment predictor variable explained respectively 44 or 39 % of the variation in the data compared to 46 % that was explained by the best overall model with SIS. For EPT richness and % EPT the most highly ranked models of sediment cover or depth explained respectively 29 or 28 % compared to 34 % and 45 or 44 % compared to 52 % of the variation in the data. Consequently, sediment cover performed slightly better than sediment depth for two of the three invertebrate community indices.

Models for algal taxon richness

Only three of 51 regression models were significant in the case of algal taxon richness, and the explanatory power of these was generally low (R-squared values between 0.09 and 0.12). The best and second-best models were nutrient single-stressor models, with DRP and $\log_{10}(\text{DIN})+\log_{10}(\text{DRP})$ representing the nutrient variable, respectively. The third-best model was of form 7, incorporating sediment (SIS) as a predictor variable in the interaction term. The regression parameters and R-squared of the best model for algal taxon richness are given in Table 3, and the corresponding 3-D graph is presented in Figure 1.d.

Table 3: Variables and parameters of best overall model of the form $Y = b_0 + b_1 \times N$ (Y = response variable, N = nutrient predictor variable, b_0 and b_1 = regression parameters, R^2 = R-squared value)

Y	N	b_0	b_1	R^2
Algal taxon richness	ln(DRP)	17.63	– 1.67	0.12

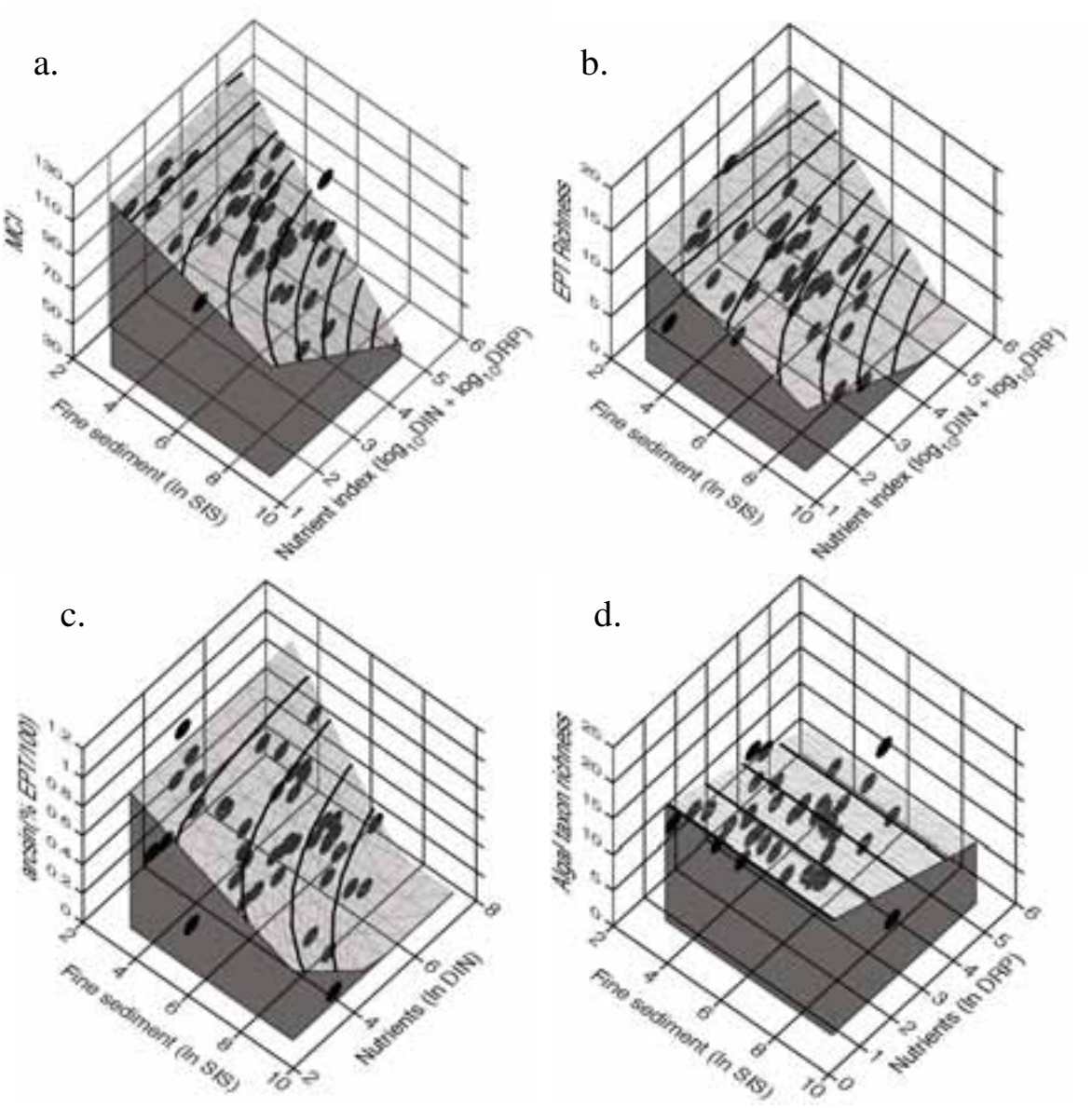

Figure 1: 3-D graphical presentations of the overall best regression models for each biological response variable along with the field survey data presented as dots. a. MCI; b. EPT richness; c. arcsin(%EPT/100); d. algal taxon richness.

Discussion

Multiple-stressor effects on stream health

Individually, both fine sediment on the streambed and nutrient concentrations in the stream water had negative effects on MCI, EPT richness and % EPT, as demonstrated by significant single-stressor models with negative coefficients. However, multiple-stressor models produced better fits to the survey data than single-stressor models for all three macroinvertebrate community indices. The fact that models incorporating both stressors could explain more variation in the data indicates that nutrients and fine sediment either affect different macroinvertebrate populations, or the same populations but through different mechanisms. The resulting change in community structure is thus the outcome of a combined multiple-stressor effect.

Multiple-stressor effects of fine sediment and nutrients on MCI, EPT richness and % EPT were of a complex (non-additive) nature, as shown by inclusion of an interaction term in the best-fitting models. Although the stressors affected communities along different pathways they did not act completely independently but interacted so that the effect of one stressor on the ecological response was dependent on the other. The best models for MCI, EPT richness and % EPT in the Southland streams included a positive relationship with nutrients (either DIN or the combination of DIN and DRP) and a negative relationship with the interaction term. At very low sediment levels, increased nutrient concentrations seem to have a small positive effect on stream health indices, but any positive effect is overwhelmed when sediments are present. Moreover, as sediment on the streambed increases the negative effect of nutrients becomes stronger. The effect of sediment on stream health, on the other hand, is negative at all nutrient levels but becomes stronger at higher nutrient concentrations. As a consequence of the negative interactive effect of the two stressors, stream health is reduced markedly when both nutrients and fine sediment levels are high.

Stressor effects could not be demonstrated for total invertebrate taxon richness. These results are consistent with a previous survey of Otago streams (Townsend et al. 2008). Since effects could be clearly shown for the richness of the sensitive EPT taxa but not for total invertebrate taxon richness, which also includes pollution-tolerant taxa, we conclude that the latter are were not sensitive to nutrient and fine sediment levels in the surveyed Southland streams and that total invertebrate taxon richness is not suitable as a health index for this region. On the other hand, % EPT (the combined number of individuals in the orders Ephemeroptera, Plecoptera and Trichoptera expressed as a percentage of the total number of individuals present) was a very

good indicator of stream health as related to the focal stressors. In fact, with 52 % of variation in the % EPT data explained by our best multiple-stressor model, % EPT seems to be the best macroinvertebrate indicator of the four investigated.

In the case of algal taxon richness, which is often used as a further stream health indicator, we found that nutrient concentration, in particular of dissolved phosphorus (DRP; note this contrasts with the patterns for macroinvertebrates where DIN was important), was a better predictor than deposited fine sediment. However, the overall explanatory power of nutrients and fine sediment was low for this biological response variable, therefore we conclude that algal taxon richness is not a promising indicator for tracking impacts of land-use intensification in Southland.

Management implications

Our results show that macroinvertebrate community indices (MCI, EPT richness and % EPT) are a function of land-use-related deposited fine sediment on the bed and nutrient status of the water in streams routinely measured by Environment Southland. These indicators should therefore be useful in tracking long-term changes in stream health related to ongoing land-use intensification or remediation efforts.

In deciding upon measures to prevent degradation or mitigate adverse land-use effects, it is important to know not just the state of the stream health indicators, but also the cause-effect relationships between stressors and stream health together with the current state of the stressors. Since cause-effect relationships between fine sediment, nutrients and macroinvertebrate community indices have been shown to be complex due to interactive multiple-stressor effects, it is crucial to take both stressors into account. For example, in some cases it might be more effective to focus on reducing sedimentation than on reducing nutrient inputs whereas in others a combination of both will be needed to achieve good stream health. Nutrient concentrations are measured at some stream health monitoring sites, when these coincide with water quality sites in Southland, but this is not always the case. Deposited fine sediment has not been routinely measured although our results show this stressor can dramatically affect macroinvertebrate communities and stream health more generally. It will be helpful if both nutrients and fine sediment are routinely measured in future.

The measurement of deposited fine sediment in streams

Of the three methods to record deposited fine sediment, the measurement of suspended inorganic sediment (SIS) using the Quorer was best at accounting for variation in the

macroinvertebrate community indices. However, SIS was only slightly superior in its explanatory power to the visual estimate of % cover or depth of sediment on the streambed.

It can be argued that the Quorer method is least subject to variation due to operator subjectivity and therefore delivers the most precise estimate. On the other hand, the Quorer method is limited to depths of less than about 30 cm (depending on the height of the Quorer) and more than about 8 cm (to be able to remove a representative sub-sample). Its use is also limited to velocities of less than about 0.5 m/sec (to prevent loss of sediment from within the Quorer) and is not suitable in substrate that is dominated by boulders or bedrock. In contrast, there are no such limitations for visual estimates of % sediment cover or measurements of sediment depth in wadeable streams, although high water turbidity may sometimes preclude using these methods.

Of more significance is the fact that the Quorer method is by far the most labour-intensive of the three methods, involving both field sampling and processing of sediment samples in the laboratory. In contrast, the visual estimation of % sediment cover and measurement of sediment depths are completed in the field. We estimated that a site measure of SIS (from 5 Quorer samples) is 7.4 times more labour intensive than a site measure of % sediment cover (from 10 visual observations). All these factors need to be taken into account when deciding on the method to use in future monitoring.

Recommendations

1. Neither total macroinvertebrate taxon richness nor algal taxon richness were strongly related to deposited fine sediment or nutrient concentrations. These biological response variables are thus likely to be less useful as indicators when tracking impacts of land-use intensification in Southland or the consequences of remediation. Total macroinvertebrate taxon richness will continue to be available because MCI and EPT indicators require complete sample identification. However, consideration can be given to discontinuing algal sampling if funding issues are paramount.

2. According to our results, % EPT is the health indicator most responsive to land-use generated changes in sediment cover and nutrient concentrations. It would be unwise only to rely on MCI and EPT estimations should be paramount in future monitoring.

3. We recommend that in future there should be routine assessment of deposited fine sediment and nutrient concentrations in SoE stream health monitoring sites. This is because (i) knowledge of the current state of fine sediments and nutrients puts managers in a better position to decide

upon the most effective mitigation measures to improve stream health, (ii) a long-term dataset will allow for further investigation of multiple-stressor effects of continued land-use intensification in Southland, (iii) a dataset collected in Southland might be used to develop sediment guidelines on a national scale and (iv) reporting sediment levels in Southland streams will raise public awareness of the issue.

4. If time and money are available, sediment on the streambed (SIS) should be assessed using the Quorer method. However, bearing in mind the high costs associated with the Quorer and the fact that visual estimation of % sediment cover accounts for only slightly less of the variation in macroinvertebrate indices, Environment Southland should consider using this more cost-effective method. An easy-to-follow, step-by-step protocol is provided in Appendix 2.

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Appendix 1: Individual site scores for nutrient concentrations, deposited fine sediment measures and macroinvertebrate and algal community indices (DRP=dissolved reactive phosphorus, DIN=dissolved inorganic nitrogen, SIS=suspendable inorganic sediment, MCI=Macroinvertebrate Community Index, EPT=Ephemeroptera, Plecoptera, Trichoptera).

Site ID	Site name	DRP (ug/L)	DIN (ug/L)	% cover of fine sediment	Sediment depth (mm)	SIS (g/m2)
9	Lill Burn at Lill Burn-Monowai Road	3	25	1.0	0.9	133
12	Aparima River u/s Dunrobin	6	24	0.3	0.1	141
13	Aparima River at Wreys Bush	8	426	1.5	0.6	123
14	Aparima River at Thornbury	8	395	5.3	1.1	150
19	Hamilton Burn at Goodall Road	7	44	0.8	0.5	223
20	Taringatura Creek at Taromaunga	25	106	13.0	3.1	480
21	Hillpoint Stream at Waikana Road	31	166	15.8	4.2	314
22	Otautau Stream at Otautau-Tuatapere Road	62	260	18.3	4.8	785
24	Oreti River at Wallacetown	5	783	0.3	0.1	476
25	Oreti River at Benmore	5	817	0.0	0.0	143
28	Cromel Stream at Selby Road	3	30	1.3	0.7	50
29	Irthing Stream at Ellis Road	9	1520	0.0	0.0	100
30	Dipton Stream at South Hillend Road	9	542	1.0	0.3	230
31	Winton Stream at Lochiel	250	969	5.5	1.9	287
32	Makarewa River at Wallacetown	7	408	2.3	1.4	209
35	Otapiri Stream at Anderson Road	35	33	10.0	1.2	440
36	Hedgehope Stream at Block Road	7	10	1.5	0.8	332
37	Silver Stream at Lora Gorage Road	33	47	29.0	5.2	417
38	Dunsdale Stream at Dunsdale Reserve	18	49	0.8	0.4	51
39	Waianiwa Creek 1 at Lornville Riverton H	28	426	32.5	4.9	1733
41	Waihopai Stream u/s Queens Drive	7	674	26.5	4.1	1770
51	Waikaia River at Waipounamu Bridge Road	45	395	0.8	0.1	214
52	Waikaia River u/s Piano Flat	7	19	0.0	0.0	59
53	Waikaka Stream at Gore	14	158	15.8	7.9	650
54	Mokoreta River at Wyndham River Road	7	405	8.8	1.4	1060
56	Mimihau Stream at Mimihau School Road	12	405	11.0	1.7	703
58	Otamita Stream at Mandeville	15	94	9.3	2.1	955
59	Waimea Stream at Mandeville	15	786	27.3	5.0	369
63	Waituna Creek at Marshall Road	7	103	96.5	46.9	3953
67	Waimatuku Stream at Lornville Riverton Hwy	16	1928	16.8	4.0	773
70	Murray Creek at Cumming Road	140	420	37.8	9.1	757
72	Trenders Creek at Hall Road	19	159	14.0	1.8	198
75	Thicket Burn at Lake Hauroko	9	21	44.8	4.8	1617
78	North Etal Stream u/s Dunrobin Valley R	9	52	4.0	1.3	361
79	Mararoa River at Mararoa Road Bridge	3	27	0.0	0.0	23
83	Makarewa River at Winton - Hedgehope Hwy	45	54	2.8	2.0	347
84	Oteramika Stream at Seaward Downs	25	74	14.8	1.8	860
86	Pig Creek at Borland Lodge	5	46	1.5	1.2	176
87	Wairaki River at Blackmount Road	7	65	0.0	0.0	27
88	Winton Stream at Benmore - Otapiri Road	14	52	100.0	79.3	10870
100	Makarewa River at King Road	27	43	15.3	3.1	423
142	Waituna Creek at Gorge Road	8	99	34.5	11.9	2107
161	Waihopai River at Waihopai Dam	11	884	1.8	0.7	476

Site ID	Site name	MCI	Invertebrate taxon richness	EPT taxon richness	% EPT	Algal taxon richness
9	Lill Burn at Lill Burn-Monowai Road	110	25	12	78	15
12	Aparima River u/s Dunrobin	130	14	8	89	15
13	Aparima River at Wreys Bush	110	21	11	67	23
14	Aparima River at Thornbury	100	17	9	9	12
19	Hamilton Burn at Goodall Road	107	18	9	74	25
20	Taringatura Creek at Taromaunga	111	17	9	44	12
21	Hillpoint Stream at Waikana Road	120	20	11	68	2
22	Otautau Stream at Otautau-Tuatapere Road	91	20	8	44	15
24	Oreti River at Wallacetown	120	20	10	30	24
25	Oreti River at Benmore	99	18	11	49	10
28	Cromel Stream at Selby Road	119	24	14	80	17
29	Irthing Stream at Ellis Road	113	24	13	65	10
30	Dipton Stream at South Hillend Road	98	20	10	57	10
31	Winton Stream at Lochiel	87	14	5	3	12
32	Makarewa River at Wallacetown	81	15	6	49	12
35	Otapiri Stream at Anderson Road	85	20	8	48	18
36	Hedgehope Stream at Block Road	85	15	7	37	19
37	Silver Stream at Lora Gorage Road	117	24	13	53	11
38	Dunsdale Stream at Dunsdale Reserve	127	29	19	85	13
39	Waianiwa Creek 1 at Lornville Riverton H	86	17	6	11	12
41	Waihopai Stream u/s Queens Drive	69	15	4	3	13
51	Waikaia River at Waipounamu Bridge Road	122	17	11	62	10
52	Waikaia River u/s Piano Flat	129	24	16	58	11
53	Waikaka Stream at Gore	87	20	8	47	16
54	Mokoreta River at Wyndham River Road	98	20	9	43	12
56	Mimihau Stream at Mimihau School Road	95	15	8	46	8
58	Otamita Stream at Mandeville	103	22	10	60	15
59	Waimea Stream at Mandeville	87	17	5	15	16
63	Waituna Creek at Marshall Road	64	10	2	5	NA
67	Waimatuku Stream at Lornville Riverton Hwy	84	21	8	18	7
70	Murray Creek at Cumming Road	79	16	4	9	7
72	Trenders Creek at Hall Road	120	22	13	86	12
75	Thicket Burn at Lake Hauroko	117	23	12	62	15
78	North Etal Stream u/s Dunrobin Valley R	125	22	14	67	8
79	Mararoa River at Mararoa Road Bridge	113	8	3	34	13
83	Makarewa River at Winton - Hedgehope Hwy	81	18	7	22	7
84	Oteramika Stream at Seaward Downs	73	15	4	43	16
86	Pig Creek at Borland Lodge	112	26	16	73	17
87	Wairaki River at Blackmount Road	121	14	10	87	11
88	Winton Stream at Benmore - Otapiri Road	86	13	5	4	9
100	Makarewa River at King Road	103	20	10	60	13
142	Waituna Creek at Gorge Road	89	11	5	9	20
161	Waihopai River at Waihopai Dam	80	18	6	7	17

Appendix 2:

Field protocol – Deposited fine sediment in streams

Introduction: Excessive fine sediment loading from agricultural land use is one of the major causes of reduced stream health. However, the status of fine sediment is rarely recorded at stream health monitoring sites. Routine monitoring of fine sediment will help to improve the management of this stressor. This protocol is designed for regional councils to characterise the amount of deposited fine sediment in a quick and inexpensive way by visually estimating the % cover of fine sediment on the stream bed at stream health monitoring sites. Fine sediment data can be compared to biological sampling data and robust relationships between the stressor and ecological responses can ultimately be developed to improve the management of streams.

Variable measured: visual estimate of % cover of fine sediment on the stream bed

Fine sediment defined: inorganic particles of less than 2 mm in diameter

Equipment: field sheets, flagged pegs or flagging tape, viewing box with a 3 x 3 grid drawn onto the glass and each square of the grid measuring 4 cm x 4 cm, measuring tape optional

Site: stream health monitoring site

Site length: length of riffle or run sampled for stream health monitoring

Sequence of events: perform fine sediment protocol before sampling macroinvertebrates

Time required: 15-20 min

Sampling design: % cover of fine sediment is estimated at ten locations per site and averaged. Locations lie on five equidistant transects distributed down the length of the site. Estimates are taken at two random locations per transect. See Figure 1.

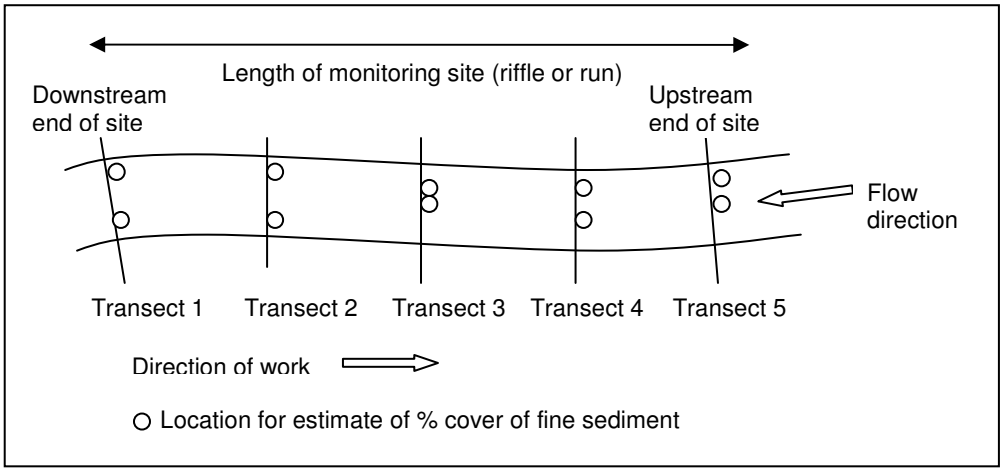

Figure 1: Sampling design for % cover of deposited fine sediment at monitoring site.

Steps:

1. Find the riffle or run where macroinvertebrates are going to be sampled for stream health monitoring.
2. Mark out five equidistant transects distributed down the whole length of the site. Use flagged pegs or flagging tape. See Figure 1.
    - An easy way to do this without the use of a measuring tape is by walking the whole length of the site and counting the steps taken. The downstream and upstream ends of the site are marked as the first and last transect, respectively. Walk back half the number of steps taken and mark the middle transect. Then again halve the distances between the middle and the upstream and downstream ends to find the last two transects.
3. Start sampling at the downstream transect. Find two random locations on the transect where % cover of fine sediment will be estimated using the viewing box. To do this divide the transect into four equal-sized sections in your mind (see Figure 2). Pick two random numbers between one and four to decide the sections to be sampled. Avoid edge habitat close to the stream bank.

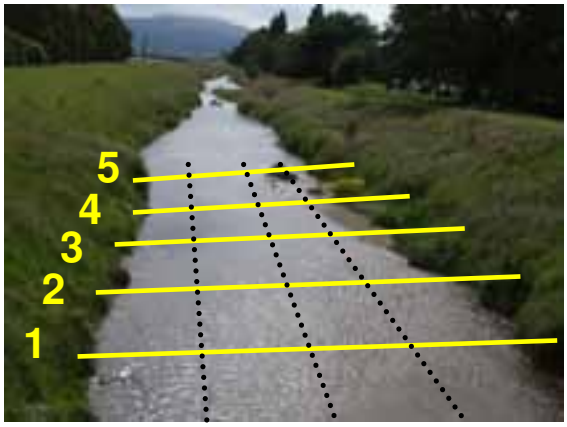

Figure 2: Five transects running across the site, each subdivided into four sections.

4. Walk along the transect to the random location. Face upstream when submersing the viewing box to avoid disturbance of the sediment during observation. Hold the viewing box firmly and don't push it too close to the stream bed, again to avoid disturbance. See Figure 3.

Figure 3: Operator estimating % cover of fine sediment of area visible through viewing box.

5. Visually estimate fine sediment cover as the percentage of the stream bed that is covered by fine sediment within the grid area that can be seen through the viewing box without moving it (see Figures 4 and 5).

    - Estimation is done in 5% increments with the exception of the value 2.5%. The value of 2.5% is assigned if a sample area is not entirely sediment-free but fine sediment does not reach a cover of 5%.
    - Using the 3 x 3 grid, estimate for each grid square the % cover of fine sediment (0%, 2.5%, 5%, 10%, 15%, …, 100%), then sum the estimates and divide by 9 (because there are 9 grid squares). This is the % cover of fine sediment for the location.
    - Only layers of fine sediment that are clearly visible with the naked eye are taken into account.

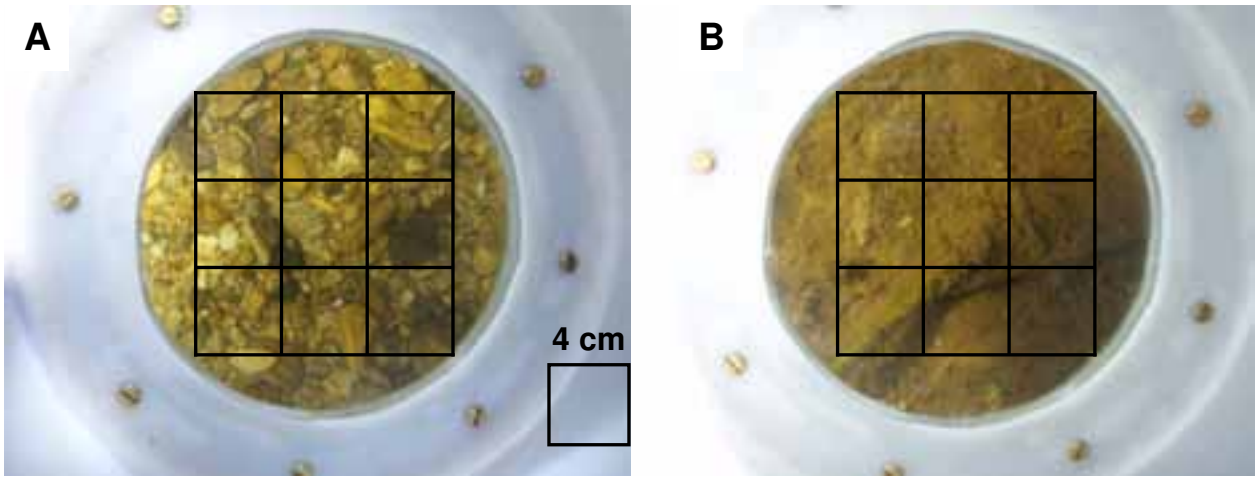

Figure 4: View of stream bed through viewing box with 3 x 3 grid. A. 0 % cover of fine sediment, B. 100% cover of fine sediment.

6. After having estimated % cover of fine sediment at two locations along the first transect move upstream to the next transect and repeat the steps.
7. The estimate of % cover of fine sediment for the site is simply the average of the ten locations. This value can be any number between 0 and 100% (i.e. not restricted to multiples of 5).

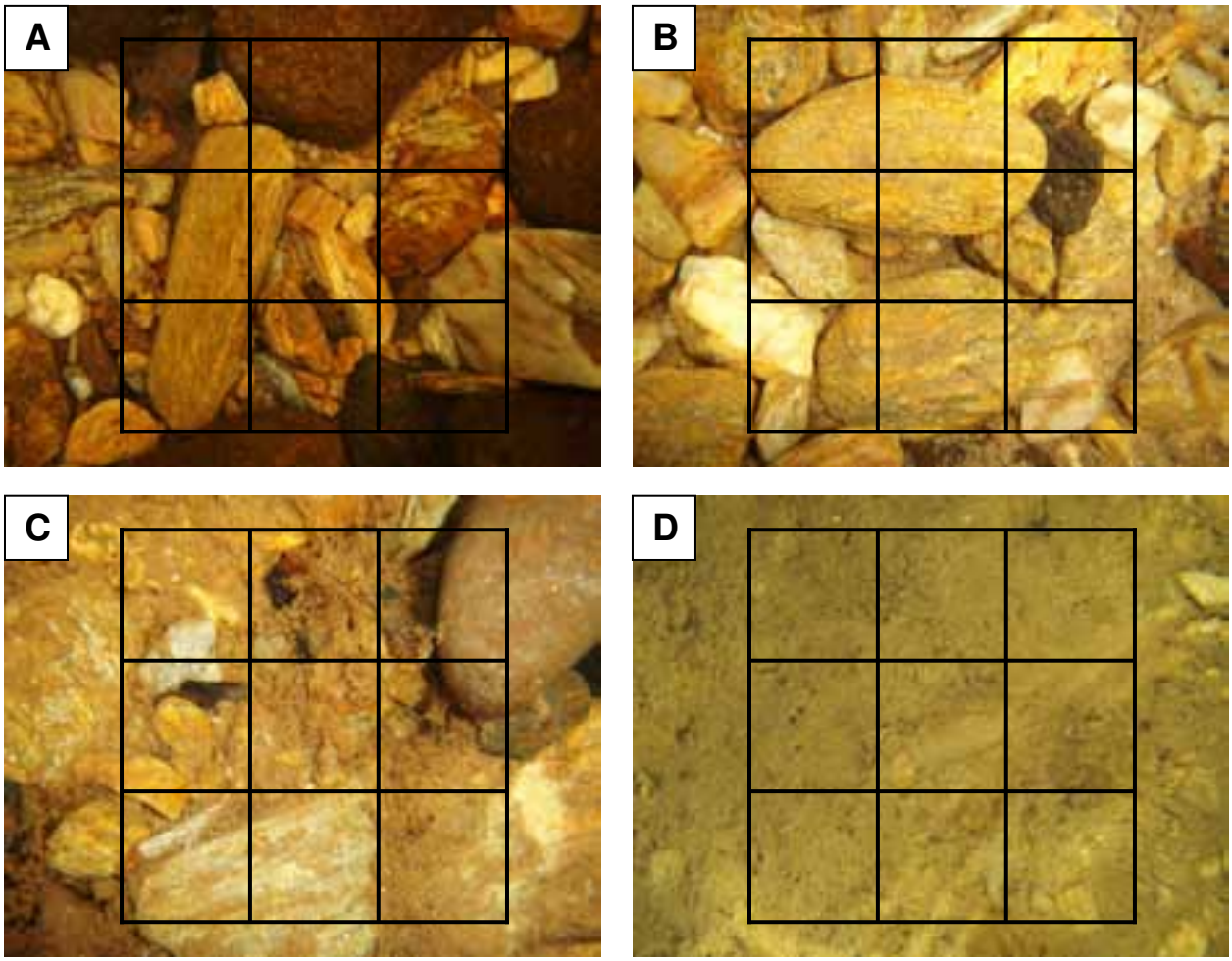

Figure 5: Examples of stream bed substrates with different percentages of deposited fine sediment cover (A. 0%, B. 5%, C. 50%, D. 100%).