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Acid Mine Drainage Index (AMDI): a benthic invertebrate biotic index for assessing coal mining impacts in New Zealand streams

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Acid mine drainage (AMD) is a widespread phenomenon globally. Drainage into streams from coal mines often contains a cocktail of acidic waters high in dissolved metals, and consequently stream invertebrate communities may be severely impacted. Traditionally, the intensity of impacts has been assessed by combinations of water chemistry and benthic invertebrate metrics; however, a metric specifically designed for assessing mining impacts has not been developed. We propose a benthic invertebrate biotic index: the Acid Mine Drainage Index (AMDI), based on species presence data. The AMDI has been developed by associating water chemistry and benthic invertebrate community data collected from 91 sites. AMD indicator scores for 57 taxa were calculated using weighted averaging. Site scores can range from 0 (severely impacted) to 100 (unimpacted) and sites can be categorised as 'severely impacted', 'impacted' or 'unimpacted'. Comparisons between AMDI and traditional indices indicated the AMDI is more accurate at detecting mine drainage.

Keywords: biotic index; acid mine drainage; benthic invertebrates; biomonitoring; stream health; coal mining

Introduction

Stream biota responses to acid mine discharges associated with past and present coal mining have been studied extensively worldwide (Verb & Vis 2000; Garcia-Criado et al. 2002; Niyogi et al. 2002; Pond et al. 2008; van Damme et al. 2008), and are the subject of continuing research in New Zealand (Winterbourn & Ryan 1994; Winterbourn & McDiffett 1996; Winterbourn 1998; Harding & Boothroyd 2004; Bray et al. 2008; Greig et al. 2010, Hogsden & Harding 2011). Since the 1850s, significant areas of the West Coast of the South Island of New Zealand have been mined for coal. Historic and current mining, especially on the Stockton–Denniston Plateau, within the Rapahoe region north of Greymouth and near Reefton, have bequeathed a legacy of water quality issues and impaired ecological communities.

Significant coal deposits in this region occur within a formation known as 'Brunner Coal Measures', these coal seams are relatively high in sulphur and metals (e.g. iron, aluminium, nickel, arsenic and zinc), and mining of these deposits frequently results in extreme contamination of downstream lotic systems (Harding & Boothroyd 2004). In severe cases, the pH of receiving waters is < 3 and the concentrations of iron, aluminium, arsenic and sulphur may be high (Winterbourn 1998). Furthermore, metal hydroxides (primarily iron) may form dense precipitates on stream substrata, which clogs stream interstices and reduces the food resource

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of grazing invertebrates. Benthic invertebrates may be severely affected by acidity and metal toxicity derived from acid mine drainage (AMD) and precipitates, and are typically reduced greatly in diversity and abundance (Winterbourn 1998; Harbrow 2001; Niyogi et al. 2002; Bradley 2003; Barnden 2005; Kitto 2009). Where pH and metal concentrations are less severe, chronic toxicity may still impair growth and development of invertebrate populations (Harding & Boothroyd 2004). However, studies of streams affected by AMD on the West Coast have reported the presence of a number of taxa more usually associated with unpolluted or 'natural' streams in waters with low pH because of high concentrations of organic acids (Winterbourn 1998; Suren & McKerchar 2001; Harding 2002). The tolerance of some taxa, such as the stonefly *Spaniocercoides philpotti* and caddisflies *Psilochorema* sp., to naturally acidic conditions in brownwater streams, high in dissolved organic carbon, may partially explain this observation (Collier et al. 1990; O'Halloran et al. 2008).

Benthic invertebrates are used widely as biological monitoring tools to detect and assess the degree of human impacts on freshwater systems (Rosenberg & Resh 1993). The popularity of benthic invertebrate biomonitoring tools has led to a proliferation of analytical approaches. Common approaches include the use of indicator species, biotic indices, and multi-metric and multivariate predictive model approaches (Stark 1985; Rosenberg & Resh 1993; Stark 1998; Karr 1999; Hawkins et al. 2000). However, the advantages and disadvantages of these different approaches continue to be the source of vigorous debate (Karr 1999; Hawkins et al. 2000), and despite polarisation of views by some workers, it is apparent that no single approach may be applicable to all types of pollution or ecosystems.

Within New Zealand, several biotic indices and predictive models have been developed and assessed to measure the health of waterways (Stark 1985, 1998; Suren et al. 1998; Joy & Death 2003; Wright-Stow & Winterbourn 2003; Stark & Maxted 2007b). The most widely adopted of the biotic indices is the Macroinvertebrate Community Index (MCI), which was originally constructed using data collected from agricultural streams in the Taranaki region (Stark 1985, 1998). The basic calculation for the MCI involves summing the tolerance values of taxa present and dividing this total by species richness. Tolerance values were determined initially by a weighting procedure based on the relative percentage occurrence of taxa at three site groups differing in their enrichment status (i.e. clean and un-enriched, slight to moderate pollution, moderate to gross pollution) (Stark 1985). Tolerance values for less common taxa, for which this procedure was unreliable (Stark 1985), or those added subsequently (Stark 1993, 1998) were assigned by professional judgement. The MCI, and its quantitative variants, have subsequently been applied to assess the effects of a wide range of land-use activities including agriculture, forestry, urbanisation and mining (Storey & Cowley 1997; Hickey & Clements 1998; Harding et al. 2000; Hall et al. 2001; McMurtrie & Taylor 2003). Other commonly used indices assess impacts by comparing the relative richness or abundance of mayflies, stoneflies and caddisflies (EPT) to the rest of the community or the richness and abundance of the total invertebrate community. EPT taxa generally require clean water and often are the first taxa to be lost from a stream system when water quality declines. The efficacy of total taxonomic richness as an indicator of stream health relies on the commensurate loss of taxa as the severity of impacts increase.

However, whilst potentially informative, the use of generic indices can produce misleading results. Comparisons of the relative number of EPT taxa are of limited use in assessing AMD impacted streams because of the tolerance of several New Zealand stonefly and caddisfly taxa to lower pH and elevated metal concentrations (Winterbourn & McDiffett

1996; Hickey & Clements 1998; Winterbourn 1998). Total taxonomic richness may also provide unreliable information if clean water taxa are replaced by equal or greater numbers of those that prefer the conditions provided by an impact, although this may not commonly occur in AMD streams. Impact specific biotic indices, such as the MCI, were developed partially to overcome these issues. However, the MCI in reality ought only to be applied in organically polluted streams, the situation for which it was designed. The use of the MCI in freshwater ecology in New Zealand over the last 20 years has highlighted both its strength as a relatively robust tool for assessing agricultural impacts, and its weakness as a ubiquitous index for measuring many other kinds of human impacts. In particular, the MCI can give highly variable and unrealistic results for stream systems affected by AMD primarily because of AMD tolerance in otherwise clean water taxa (Harding et al. 2000). Several overseas studies have suggested the use of abundance metrics (Schmidt et al. 2011); however, these should be applied cautiously in streams with high physical disturbance where flooding has large impacts on abundance, such as those on the West Coast of the South Island. Quantitative abundance metrics also demand costly degrees of replication to overcome spatial variability (Boothroyd & Stark 2000).

The absence of an index or assessment tool suitable for AMD impacts on New Zealand's streams has hindered the interpretation and communication of findings to non-technical audiences. Thus, our aim was to develop the Acid Mine Drainage index (AMDI), a biotic index specifically sensitive to the effects of mine drainage chemical pollution in New Zealand streams.

Methods

The development of the AMDI index involved two stages; first, we identified water chemistry parameters that were significantly correlated with gradients in community composition using multivariate ordination techniques. Secondly, using those water chemistry parameters and weighted averaging (WA) techniques we established taxa specific indicator scores. Whilst we considered the development of other biomonitoring techniques, specifically predictive modelling, the large number of sites required for these methods relative to the number and variety of AMD streams available on the West Coast prevented this technique being applied (Jimenez-Valverde et al. 2009).

Study sites

We used data collected from surveys over 4 years from 91 separate sampling locations on the West Coast of New Zealand (Fig. 1). All sites fall within the Westland Forest Ecoregion (Harding & Winterbourn 1997). A range of physico-chemical parameters and benthic invertebrate communities were sampled during periods of summer base flow between 2003 and 2007. The primary source of data for development of the index came from streams located on the Stockton–Denniston Plateau (north of Westport). Brunner coal capped by Kaiata mudstones and granite has been mined in this area since the 1850s. The plateau is approximately 700 m a.s.l and receives relatively high rainfall ($>$3000 mm p.a.). Many streams on the plateau are strongly incised, cutting through small canyons 10–20 m deep within podocarp forest riparian zones. Streambeds are often dominated by bedrock substrate while boulders and cobbles are less abundant. Additional AMD impacted and putative reference streams were located around the township of Reefton (upper Inangahua River area), which has a lower rainfall than the plateau streams and mixed beech–podocarp forested hill catchments. Streams near Reefton were dominated by boulder, cobble and gravel substrates. The remaining streams were located within the Grey and Taramakau river valleys close to the town of Greymouth (Fig. 1). Streams there were amongst mixed

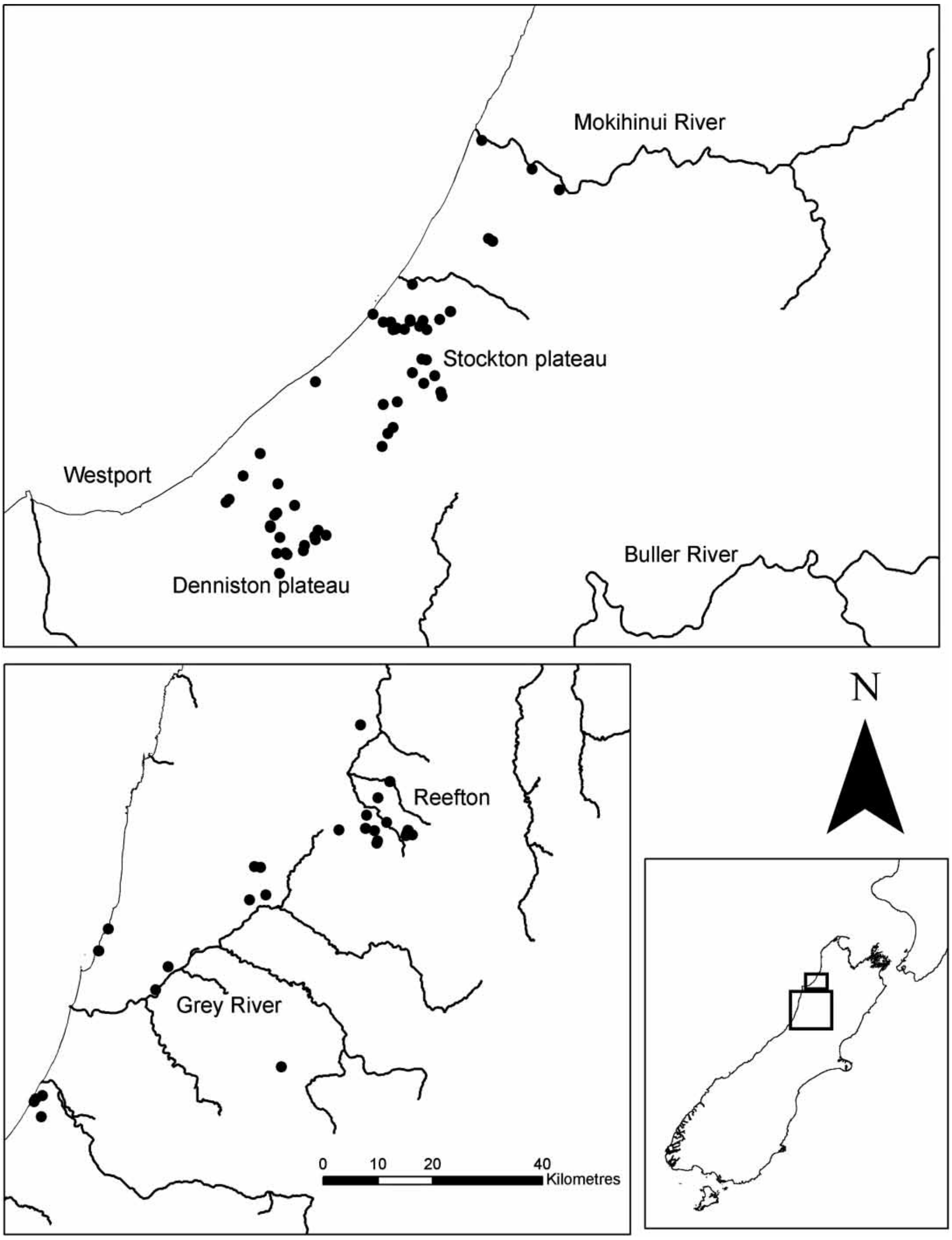

Figure 1 Location of 91 streams on the West Coast of the South Island sampled for development of the Acid Mine Drainage Index (AMDI).

podocarp forest and dominated by bedrock, boulder and cobble substrates. The majority of accessible and wadeable AMD impacted streams on the West Coast were sampled.

Field data

Water chemistry parameters were measured within a 50-m reach at each stream. Spot measurements of specific conductivity, pH and temperature (YSI Model 63 meter), dissolved oxygen (YSI 550DO meter) and turbidity (HACH 2100P turbidimeter) were made in the field. Water grab samples (250 ml) were collected in acid-washed bottles from mid-stream and mid-depth, kept cool and returned to the laboratory for analysis. In the laboratory, samples were filtered through 0.45-μm membranes and preserved with nitric acid. Samples were then analysed for dissolved iron (Fe), aluminium (Al) and nickel (Ni), using an ICPMS (Hill Laboratories Ltd., Hamilton). Detection levels were 0.02 mg/l for iron, 0.003 mg/l for aluminium and 0.0005 mg/l for nickel. Where quantities were found to be below the detection limit, values at half that detection limit were used in analyses. Hill Laboratories is an accredited (IANZ) laboratory. Quality control consisted of procedural blanks, sample spikes, duplicate and repeat samples. All analyses were referenced to APHA 3125 B 21st edition 2005, and regular external checks are made through participation in the Inter Laboratory Comparisons Program (ILCP).

Stream disturbance was assessed using the Stream Channel Inventory (Pfankuch 1975). Stream width (m) and depth (m) were recorded at each site and the dimensions of 30 particles measured and used to calculate a substrate index (Jowett & Richardson 1990). Elevation was measured using a hand-held Garmin GPS.

At each site an extensive kick-net (0.5-mm mesh) sample was collected from a range of habitats, including riffles, pools, bedrock, boulders, moss and woody debris. Individual stones were also handpicked for benthic invertebrates. In order to develop a robust AMDI index, it was important to collect as many taxa as possible, particularly taxa most tolerant of mining activities. Therefore, sampling effort was greater at moderately and heavily impacted AMD sites where abundances of invertebrates are typically extremely low (e.g. 1–2 animals per 5 m). Samples were preserved in 70% ethanol in the field and returned to the laboratory. Samples were sorted in their entirety and identified using Winterbourn et al. (2006). For the development of the AMDI index, the same taxonomic resolution currently used in the MCI (Stark 1985) was adopted (i.e. genus, family or ordinal level).

Survey results

Environmental conditions

Physical and chemical conditions varied widely across the 91 streams (Table 1). Unimpacted stream water would typically be low in metals and conductivity, but pH might vary naturally between circum-neutral and 4.5. Overall, stream pH had a median value of 4.6, but ranged between 8 in a forested stream close to Reefton and 2.7 in upper Mine Creek at Stockton on the Stockton–Denniston Plateau. Similarly, conductivity showed a large range from 19.2 (μS_{25}/cm) in an un-impacted stream near Denniston on the Stockton–Denniston Plateau to 2846 (μS_{25}/cm) in upper Mine Creek.

Table 1 Average values ($\pm$1SE) and the range of environmental variables used in partial Canonical Correspondence Analysis (pCCA) analysis of acid mine drainage (AMD) streams.

Variable	Mean	±SE	Max	Min
AMD variables				
pH	4.6*	NA	8.0	2.7
Conductivity (μS_{25}/cm)	262.2	44.1	2846.0	19.2
Fe (μg/l)	3304	1119	84500	20
Al (μg/l)	6512	2089	159000	17
Ni (μg/l)	26.0	7.1	302.0	0.5
CCU	46.9	14.9	1146.3	0.2
Environmental co-variables				
Elevation (m)	339.7	26.1	900.0	0.0
Pfankuch (1975)	70.1	2.3	126.0	38
Width (m)	3.2	0.3	15.0	0.2
Depth (m)	0.1	0.00	0.5	0.01
SI	5.7	0.1	8.0	0.0

*pH is presented as a median value because it is not appropriate to average a variable on a logarithmic scale. CCU, cumulative criterion unit; SI, Substrate Index.

AMD water typically is composed of a cocktail of metals, which might have additive effects upon biota, especially at chronic concentrations (Guasch et al. 2009). Therefore, we calculated a measure of total metal concentration and toxicity of metals: the cumulative criterion unit (CCU) (Clements et al. 2000). CCU scores were calculated as:

$$CCU = \sum \frac{m_i}{c_i}$$

where m_i is the dissolved concentration and c_i is the criterion value for the *i*th metal based on United States EPA. guidelines on critical concentrations, which when exceeded may be harmful to aquatic organisms.

Dissolved metal concentrations ranged across orders of magnitude, particularly aluminium, and this pattern was evident in the range of CCU scores (Fig. 2). When sites were ranked according to CCU score and plotted against dissolved metal concentrations, the contribution of each metal to CCU is apparent. After log-transformation to better meet the assumptions of normality, aluminium concentration had the greatest influence upon CCU (Pearson correlation $r = 0.994$), although iron ($r = 0.885$) and nickel ($r = 0.876$) were also highly influential.

The average altitude of sites was 339 m a.s.l and ranged from 0 to a maximum of 900 m a.s.l. Stream physical disturbance, primarily from flooding, was measured using the Stream Channel Inventory (Pfankuch 1975). Streams were scored between 38, very stable, and 126 (out of a maximum of 152), very disturbed. Sample reaches had an average width of 3.17 m, but ranged from 0.2 to 15 m. Depths were less variable because of the constraints of invertebrate sampling in deep water. Substrate measurements were transformed into a single variable by

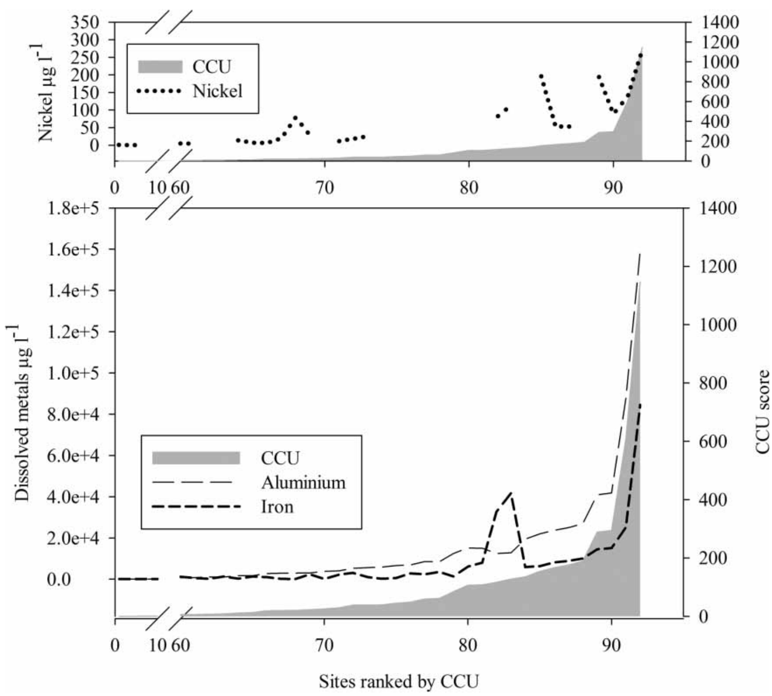

Figure 2 The concentrations of dissolved aluminium, iron and nickel at 91 sites relative to cumulative criterion unit (CCU) scores derived from them. Sites are ranked according to CCU.

summing weighted substrate percentages to form a substrate index (Jowett & Richardson 1990).

$$\begin{aligned}\text{Substrate index} = {} & \\ & 0.08\%\text{bedrock} + 0.03\%\text{boulder} \\ & + 0.06\%\text{cobble} + 0.05\%\text{gravel} \\ & + 0.04\%\text{finegravel} + 0.03\%\text{sand.}\end{aligned}$$

Substrate index scores ranged from 0, all silt, to 8, which is an entirely bedrock stream. The average stream had a substrate index score of 5.7, which is a gravel/cobble-dominated stream.

Invertebrate communities

Not surprisingly, invertebrate communities were highly variable. Nine sites contained a single taxa (sites devoid of taxa were not included during construction of indices), which were most commonly the dipteran chironomidae, but also the plecopterans *Spaniocercoides* spp. or *Zelandobius* spp. The most diverse stream contained 30 invertebrate taxa, which included members of all the main stream invertebrates typical of the West Coast ecoregion (Harding & Winterbourn 1997).

Development of the Acid Mine Drainage Index (AMDI)

We used multivariate ordinations, conducted in CANOCO version 4.55 (ter Braak & Smilauer 2006), on presence–absence data to determine the AMD variables, whilst controlling for other environmental variables, which were the most important determinants of invertebrate community composition in our study streams. Invertebrate taxa found in three or fewer sites were removed from the analysis giving a total of 57 taxa for which an indicator score could be calculated. The frequency of taxa present at varying AMD impact levels along the ordination axes allowed the identification of taxon-specific impact optima using the method of WA (ter Braak & Juggins 1993). The assignment of indicator values based upon environmental optima provides the ability to reduce community data to a linear scale of AMD impact from natural to highly polluted (Brandt 2001; Smith et al. 2007). Three is the absolute minimum number of occurrences from which to calculate a tolerance value and we concede that for taxa represented by a low number of occurrences there remains a proportionally increasing degree of uncertainty that we sampled an adequate fraction of tolerable conditions. However, many of these taxa are naturally rare and it is not possible to acquire adequate occurrences from the available selection of impacted streams to satisfy statistical requirements completely. We believe that to omit less common taxa would detract from the efficacy of this index to a greater degree than incurred by including infrequent taxa. Therefore, we opted to retain these uncommon taxa but have included the number of observations made for each in Table 2. As more streams are sampled and taxa observed it will be possible to adjust tolerance values accordingly.

Determining the mine impact gradient

Multivariate ordination techniques have the advantage of allowing for variables of interest to be tested after the removal of variation relating to other environmental conditions such as stream size, substrate or position in the landscape. Spatial autocorrelation is the tendency for adjacent sites to contain a more similar suite of taxa than those further apart by virtue of that proximity, rather than any underlying environmental factor (Lichstein et al. 2002). Thus, if the focus is environmental regulation of communities, spatial autocorrelation constitutes background noise (Borcard et al. 1992; Peres-Neto & Legendre 2010). We created spatial predictor variables for our data set using principal coordinates of neighbour matrices (PCNM) in R 2.13.0 (R core development group 2011) using the package ‘PCNM 2.1’ (Dray et al. 2006). We also wished to isolate the effects of AMD on stream invertebrate communities from other determinants

Table 2 Indicator scores for the cumulative criterion unit (CCU), Acid Mine Drainage Index (AMDI), CCU optima and maximums for each taxa and the number of observed occurrences upon which optimum values were based.

	Indicator score	CCU optima	CCU max	No. observations
Ephemeroptera				
Ameletopsis	6	1.94	5.2	6
Austroclima	4	2.65	5.2	4
Coloburiscus	8	1.33	5.2	28
Deleatidium	6	1.97	22.9	53
Ichthybotus	10	0.77	1.2	4
Neozephlebia	3	3.30	38.6	20
Nesameletus	9	1.02	2.6	14
Zephlebia	9	1.19	3.0	13
Plecoptera				
Austroperla	0	12.56	108.0	20
Cristaperla	9	0.98	2.2	4
Megaleptoperla	9	1.17	1.3	4
Spaniocerca	4	2.63	25.4	12
Spaniocercoides	1	9.79	38.6	17
Stenoperla	7	1.67	5.2	20
Taraperla	10	0.94	1.5	7
Zelandobius	0	12.76	1146.3	35
Zelandoperla	4	2.29	19.0	27
Trichoptera				
Alloecentrella	3	2.97	4.9	5
Hydropsychidae	5	2.26	20.3	25
Costachorema	9	1.18	2.2	8
Helicopsyche	8	1.48	3.9	9
Hudsonema	6	1.99	5.2	7
Hydrobiosella	7	1.74	5.2	13
Hydrobiosis	2	3.96	39.2	31
Hydrochorema	2	3.91	22.9	6
Oeconesus	2	6.30	19.0	4
Olinga	10	0.92	3.9	16
Oxyethira	1	11.42	106.9	23
Plectrocnemia	8	1.24	2.1	4
Polyplectropus	4	2.79	19.0	13
Psilochorema	2	4.70	48.8	19
Pycnocentria	7	1.83	5.2	19
Pycnocentrodes	0	35.55	608.7	11
Rakiura	7	1.71	5.2	18
Triplectides	6	1.95	5.2	11
Zelolessica	3	3.34	5.2	6
Diptera				
Aphrophila	8	1.66	4.9	13
Austrosimulium	5	2.19	5.2	14
Ceratopogonidae	5	2.04	8.3	6
Chironomidae	0	47.01	608.7	79

Table 2 (*Continued*)

	Indicator score	CCU optima	CCU max	No. observations
Empididae	5	2.10	5.2	11
Eriopterini	3	3.41	59.5	19
Limonia	7	1.66	3.9	10
Paralimnophila	6	1.91	8.3	11
Muscidae	0	95.94	166.5	6
Coleoptera				
Hydora (Elmidae)	0	16.74	189.3	35
Zeahydora (Elmidae)	9	1.15	3.0	5
Hydraenidae	5	2.05	4.9	10
Hydrophilidae	8	1.66	4.6	8
Ptilodactylidae	10	0.83	1.3	4
Scirtidae	1	7.13	59.5	27
Other				
Archichauliodes	2	3.93	20.3	23
Oligochaeta	1	7.40	106.9	39
Ostracoda	4	2.45	5.9	5
Paraleptamphopus	3	3.82	19.0	9
Paranethrops	10	0.81	1.8	5
Potamopyrgus	10	0.71	1.2	11

such as stream disturbance, size and substrate composition. These co- or conditioning variables were combined with the PCNM spatial vectors in a partial ordination, which allowed only the influence of AMD chemical gradients to influence ordination axes.

Rare taxa were down weighted in all ordination analyses. An initial Detrended Correspondence Analysis (DCA) of invertebrate communities showed that the longest gradient length was 3.963; therefore taxa responses were unimodal and Canonical Correspondence Analysis (CCA) was the most appropriate model to apply to these data (Leps & Smilauer 2003). All pairwise combinations of variables (i.e. 51 PCNM vectors and five physical factors, the AMD water chemistry variables pH, CCU and conductivity) were checked for collinearity (Table 3) in R version 2.13.0 (R core development group 2011). However, none was highly correlated ($r < 0.9$; Quinn & Keough 2002), so all variables were left in the

Table 3 Pearson product moment correlation matrix between acid mine drainage (AMD) and environmental co-variables used in partial Canonical Correspondence Analysis (pCCA).

Conductivity	−0.413						
CCU	−0.781	0.747					
Elevation	−0.399	−0.136	0.055				
Pfankuch	0.473	−0.145	−0.279	−0.504			
Width	0.123	0.059	0.143	−0.406	0.403		
Depth	0.489	−0.134	−0.204	−0.578	0.69	0.532	
SI	−0.111	−0.055	0.009	0.139	−0.378	0.093	−0.215
	pH	Conductivity	CCU	Elevation	Pfankuch	Width	Depth

CCU, cumulative criterion unit; SI, Substrate Index.

subsequent analyses. Specific conductivity (μS_{25}/cm), CCU, elevation (m) and width were log-transformed prior to analyses to achieve normal distributions. Using CCA, manual forward selection and significance testing based on 999 Monte Carlo permutations we selected a suite of significant ($p < 0.05$) spatial and environmental co-variables. Finally, we performed a partial Canonical Correspondence Analysis (pCCA) using the significant co-variables and carried out a further manual forward selection on the AMD water chemistry variables to detect significant relationships. During CCA analyses, we used Hills scaling focused on inter-sample distances (Leps & Smilauer 2003) and interpreted only the first two axes.

Weighted averaging (WA)

To develop indicator scores we used a similar method to Brandt (2001) and Smith et al. (2007). This involved estimation of weighted averages for each taxon using water chemistry variables, selected by pCCA and allocated to one of 13 ranges (bins), which contained roughly equal numbers. The optimum value of a taxon along the gradient of impact was calculated by dividing the sum of the weighted proportion of times a taxon occurred within the 13 bins, by the sum of the un-weighted proportion of times a taxon occurred within the bins.

$$\textit{Environmental optima} = \frac{\sum (Wprop)_{bin1=bin2=....bin13}}{\sum (Uprop)_{bin1=bin2=....bin13}}$$

where *Wprop* is the mean water chemistry value of each bin, multiplied by the proportion of time the taxon occurred within each bin (*Uprop*). Thus, the environmental optimum of each taxon is roughly equivalent to the mean water chemistry value of the bin in which the taxon had the greatest number of occurrences.

Indicator values were then assigned on the basis of the water chemistry optima. Optimum values were sorted from lowest to highest (or vice versa depending on the relationship of the variable to AMD impact) and separated into relatively evenly divided groups (in this case 11 groups). Each group consisted of five taxa except the first and last groups, which had six taxa. Taxa in Group 11, which were associated with unimpacted stream water chemistry were allocated an indicator score of 10, whilst those in Group 1 (severely impacted) were allocated a score of 0. Thus, a score of 10 indicates a taxon intolerant to AMD, whilst taxa scoring 0 are highly tolerant (Table 2).

Using indicator values and presence/absence data from any site, it is then possible to calculate the AMDI:

$$\text{AMDI}\, \textit{score} = \left\{ \left(\frac{\sum b_i}{c} \right) * \log_{10} \mathrm{c} \right\} * 10$$

where b is a taxon indicator value for the ith *taxa* and c is the total number of scoring taxa found in the samples. We incorporated a richness multiplier into the formula because richness is known to be a useful indicator of AMD impacts and to prevent the occurrence of stray, low tolerance taxa overly biasing the site index score. AMDI scores range from 0 to 100, with a higher score indicating a lesser degree of AMD impact. Un-scored taxa should be left out of the index calculation.

Linear regression was used to investigate the relationships between indices calculated on different aspects of the AMD chemical impact gradient (pH or CCU), taxonomic richness and the gradient in that chemical variable. Richness and CCU were log-transformed prior to analyses to meet the assumptions of normality better. These analyses were performed using R version 2.13.0 (R core development team 2011), and informed the choice of AMD gradient, which produced the most effective AMDI index.

In order to distinguish a scale of AMD impact, we used the original series of CCU bins to represent impact classes (Smith et al. 2007). The CCU values of each bin were averaged and taxa were represented by the number of occurrences in that bin. We applied the

Bray–Curtis distance measure and performed a cluster analysis on the inter-bin matrix using the program PRIMER 6.1.12 (Clarke & Gorley 2006). The cluster analysis was then used to inform the choice of impact thresholds.

Finally, we compared the AMDI index for each of our study streams with three other biotic indices commonly used to describe macroinvertebrate communities in New Zealand: percentage EPT, EPT richness and the MCI. This was done to evaluate the relative sensitivity of various indices to AMD and check for the misclassification of streams by generic indices.

Index calculation results

Our initial CCA and forward selection of the co-variables identified significant spatial autocorrelation in the data set. Furthermore, stream depth was shown to significantly influence stream communities, and so to account for these variables PCNM vector 26 (an orthogonal spatial predictor variable) and depth were included as co-variables in the final pCCA. The pCCA analysis indicated that CCU and pH were both significant ($p < 0.05$) drivers of the gradient in community composition (Fig. 3). Therefore, we created two separate indices for CCU and pH using WA.

Although pH and CCU were correlated (Pearson's $r = 0.781$), the indicator scores derived for each index were not (Pearson's $r = 0.159$), indicating that taxa had a different response to each variable. The relationships between the CCU and pH indices, taxonomic richness and the water chemistry parameters from which they were generated are shown in Fig. 3. Both indices had a strong relationship to log taxonomic richness (Fig. 4). In the absence of further streams with which to test the efficacy of our index, we considered this an important validation of these indices as taxonomic richness is consistently low in mine-impacted ecosystems (Winterbourn 1998; Harding et al. 2000). However, despite being significant in both cases, the relationship between the indices and their respective water chemistry variable was much stronger for CCU. This weaker result for pH is most likely to be driven by the occurrence of naturally acidic streams alongside AMD and circum-neutral streams. Consequently, streams with pH as low as 3.5 had high pH AMDI scores. Therefore, we decided that the CCU AMDI was the more effective of the two indices and the pH AMDI was not considered further.

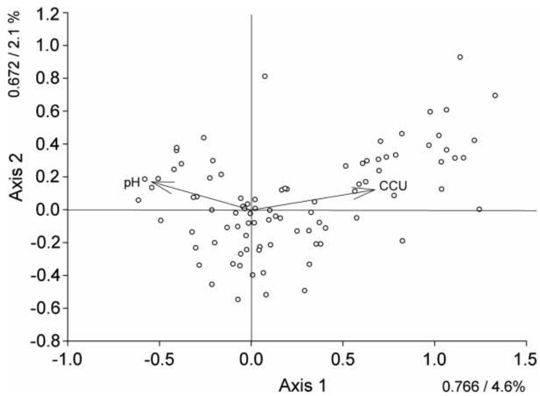

Figure 3 Partial Canonical Correspondence Analysis (pCCA) of benthic invertebrate presence–absence in 91 streams showing the relationship between community composition and significant water chemistry parameters after conditioning for the effect of stream depth and spatial autocorrelation. Species–environment correlations and % variance explained for each axis are shown. Correlations between logCCU (cumulative criterion unit), Axis 1 and Axis 2 were 0.69 and 0.28; pH, Axis 1 and Axis 2 were 0.74 and 0.17, respectively.

CCU values ranged from 0.2 to 1146.3; however, taxa CCU optima ranged between 0.71 and 95.94 (Table 2). Taxa CCU maximums ranged between 1.2 and 1146.3, although it seems unlikely that the single *Zelandobius* sp. individual found in upper Mine Creek was part of a sustainable population at this location.

Cluster analysis of bins based on mean CCU, occurrence of each taxon in that bin, and the Bray–Curtis distance measure clearly separated bins into three groupings (Fig. 5; Table 4). Bins 1–4 were the most severely

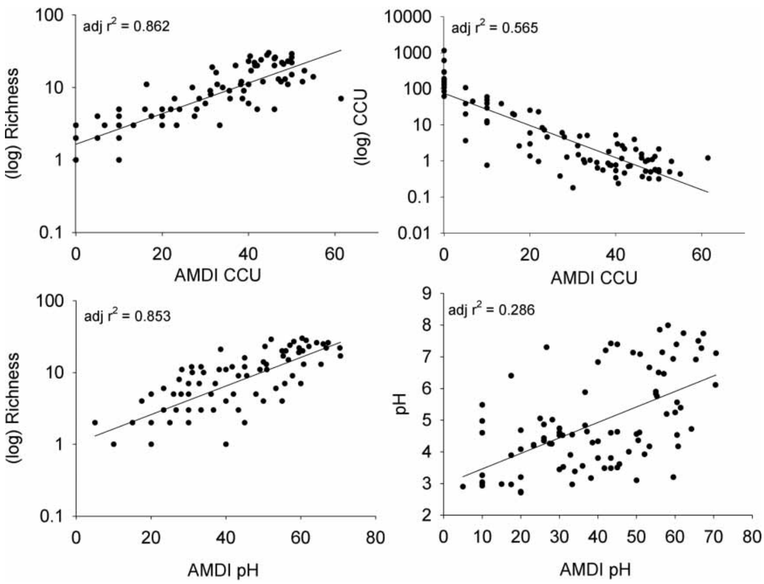

Figure 4 Relationships between the cumulative criterion unit (CCU) and pH derived indices, taxonomic richness, CCU and pH.

impacted sites, whilst bins 5–7 were moderately impacted sites and bins 8–13 were unimpacted, containing the natural reference sites. AMDI values of sites within each impact grouping showed distinct differences between groups, particularly the severely impacted sites (overall $p > 0.001$). However, although significantly different (Tukey's HSD $p = 0.032$), there was some overlap between the impacted and unimpacted streams (Fig. 6).

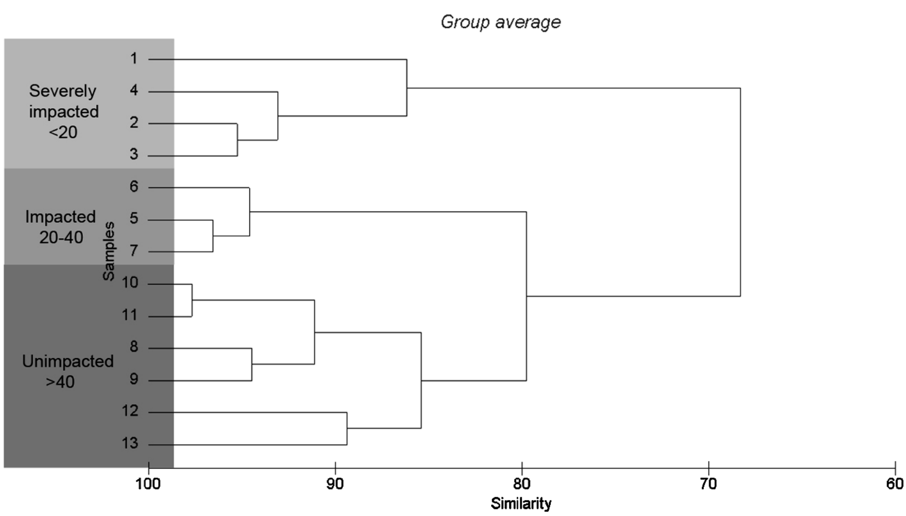

Figure 5 Cluster analysis of bins based on mean cumulative criterion unit (CCU), occurrence of each taxa in a bin and the Bray–Curtis distance measure. Impact thresholds and Acid Mine Drainage Index (AMDI) score ranges are shown.

Table 4 Likely degree of acid mine drainage (AMD) impact indicated by Acid Mine Drainage Index (AMDI) scores derived from cluster analysis.

Likely water quality	AMDI range
Severely impacted	<20
Impacted	20–40
Unimpacted	>40

Site scores range from a theoretical maximum of 100 to minimum of 0.

Finally, we compared the AMDI index for each of our study streams with three other biotic indices commonly used to describe macroinvertebrate communities (Fig. 7). The relative number of EPT taxa (or %EPT), showed considerable spread in values at low AMDI scores. Those sites which received a high %EPT score but low AMDI score can be considered as being misclassified. EPT richness showed a strong positive relationship to AMDI scores, similar to that shown by total taxa richness (Fig. 3) and no sites were considered misclassified. A comparison between the AMDI and MCI showed that the MCI misclassified several sites (Fig. 8). Impact or quality classes derived from each index are shown and 20 sites rated excellent or good by the MCI are considered impacted by the AMDI. Of a total of 35 sites classified as severely impacted by the AMDI, 57% were misclassified by the MCI. Overall, the MCI tends to over-estimate the health of macroinvertebrate communities in AMD impacted streams.

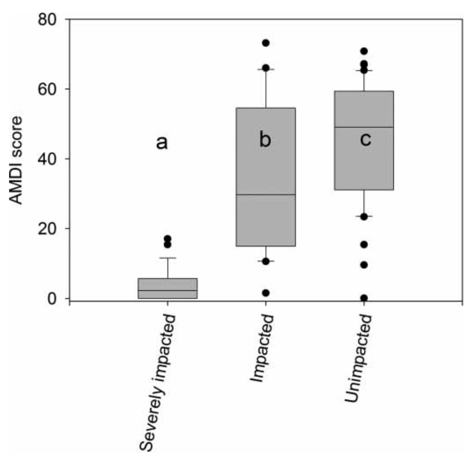

Figure 6 Acid Mine Drainage Index (AMDI) scores for the 91 streams divided into their appropriate impact category. Plots show median, 75th and 95th percentiles plus outliers. Significant differences between groups after analysis of variance (ANOVA) are indicated by letters on the plot.

Discussion

Human impacts on landscapes strongly influence the biological communities within them, but the specific effects of these changes vary across natural gradients of environmental variables and the resilience of individual taxa (Allan 2004). The AMDI index provides a specific tool for the assessment of historic and current coal mining impacts, and the monitoring of new mining developments, which allows for taxa specific adaptations to stressors in a way that more generic indices do not. Given the increased interest in mineral extraction in New Zealand from previously un-mined catchments, the development of this index is particularly timely.

The advantage of the AMDI over more generic indices such as richness, MCI or %EPT lies in a specific sensitivity to the invertebrate communities and environmental stressors found in West Coast streams. Several overseas studies suggest that abundance indices provide more useful metrics for detecting metal pollution (Schmidt et al. 2011). However, in these sites, severely impacted streams tended to have extremely low densities of invertebrates, making abundance based comparisons difficult. Typically, other indices misclassified AMD impacts because several New Zealand stonefly and caddisfly taxa are intolerant of many pollutants from common land use activities (e.g. agriculture and urbanisation), but cope with either low pH and/or high dissolved metals. For example, the stoneflies *Austroperla*, *Spaniocercoides* and *Zelandobius* all had AMDI indicator scores of <2 (indicating tolerance to

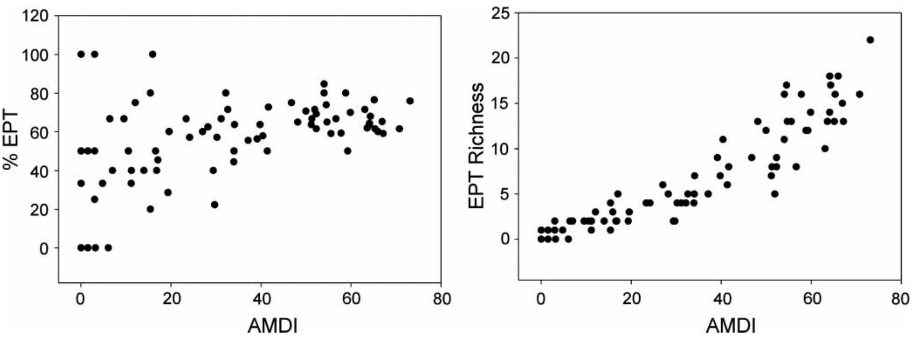

Figure 7 Comparisons between the Acid Mine Drainage Index (AMDI),%EPT taxa (mayflies, stoneflies and caddisflies) and the richness of EPT taxa for the 91 streams used to develop the AMDI.

AMD), and the caddisflies *Psilochorema*, *Oxyethira* and *Pycnocentrodes* also scored low. Consequently, among the 28 sites classified as severely impacted by the AMDI (i.e. an AMDI site score of <20), there was a range in MCI score from 0 to 160; suggesting a range in water quality from poor to excellent (Stark & Maxted 2007a). Overall, the MCI appears to have a tendency to over-estimate the health of macroinvertebrate communities in AMD impacted streams compared with the AMDI. Similarly, the ability of the taxa listed above to tolerate naturally low pH waters means that

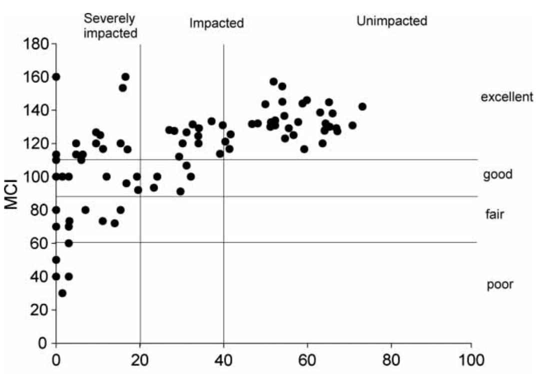

Figure 8 Comparison between the Acid Mine Drainage Index (AMDI) and Macroinvertebrate Community Index (MCI) showing the water quality or impact classes predicted by indices (Stark & Maxted 2007a).

%EPT metrics can be misleading. Several of the taxa able to tolerate the impacts of AMD are EPT, which is not the case when other impacts such as organic or thermal pollution are considered. Taxonomic richness has previously been shown to be an effective indicator of AMD impacts (Winterbourn 1998) and our results suggest that this is a valid assumption. The difficulty with using taxonomic richness alone is that other naturally occurring confounding factors, such as flood disturbance can also reduce richness. Furthermore, the richness of EPT taxa also appears to be a useful indicator of AMD impact. Therefore, EPT richness (but not %EPT) and total richness may be useful companion metrics to the AMDI. As a general rule, we would encourage the reporting of multiple metrics when determining any impacts.

Using the AMDI, three apparently 'non-AMD' sites (based on water chemistry and local knowledge) from our survey were misclassified and received an AMDI score of <20, suggesting severe AMD impacts (Fig. 6). All three were small, naturally acidic streams on the Stockton plateau (average pH 4.5) with low dissolved metal concentrations. One stream (AMDI site score = 0) contained a single taxon, the stonefly *Spaniocercoides* sp., which is known to be tolerant of low pH (Collier et al. 1990; O'Halloran et al. 2008). Two other streams had greater richness, three and four taxa respectively, and both streams contained *Deleatidium* spp., which is generally indicative of un-impacted streams (AMDI taxon indicator score 6). These streams received a low AMDI site score partially because of the taxa identified (*Spaniocercoides* taxon indicator score = 1), but primarily because of the low taxonomic richness. Low richness can be the result of a number of factors that are not immediately obvious. Firstly, all three streams are surrounded by intensive mining operations, thus reducing the opportunity for these streams to be colonised by aquatic invertebrate species. Secondly, it is possible that these streams are ephemeral or that recent high flows had altered community structure. Alternatively, peak metal concentrations may have been missed by sampling at base flows and/or on a single occasion. Irrespective of the actual cause, it is important to note that the efficacy of any biotic index will be affected by episodic natural or anthropogenic changes in environmental conditions, which may not be detected during a survey. Consequently, it is important that major environmental variables, in this case metal concentrations and pH, are collected alongside invertebrate samples and the value of temporally repeated sampling cannot be over stated.

Unfortunately, the rarity of some taxa meant that tolerance optimum and range could not be accurately determined for these organisms (and may require revision for others; Table 2). For example, the caddisfly *Kokiria* was only collected at two sites, one of which was an AMD impacted stream with pH < 3.5 and high dissolved metals; thus *Kokiria* might prove a useful species to further improve the resolution of this index. However, the absence of sufficient records (more than three) meant that a tolerance optimum could not be calculated. However, as more surveys of AMD stream systems are conducted, it will be possible to attribute scores to more taxa and increase the sensitivity of the AMDI.

We believe that we have accumulated the most comprehensive benthic invertebrate dataset of AMD impacted streams in New Zealand. The West Coast of the South Island is the major area of AMD impact and this index was specifically designed with those streams in mind. Comparisons with other stream invertebrate indices commonly used in New Zealand showed that the AMDI provides more realistic results than either %EPT or the MCI, and that EPT richness and total richness provide useful companion measures of AMD impacts. We suggest that the AMDI will provide a useful first tool for the assessment and management of AMD impacts on streams throughout New Zealand's expanding mining estate.

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