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Measuring community response of bentic macroinvertebrates in an erosional river impacted by acid mine drainage by use of a simple model

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ABSTRACT

Acid mine drainage (AMD) causes different responses in riverine benthic macroinvertebrate communities than that caused by organic pollution. The response is similar to that for metal toxicity and acidity where the impact is severe, or for inert solids where the impact is moderate to mild. Biotic indices are based on saprobity and so do not accurately reflect community disturbance for either toxicity or inert solids and thus cannot be considered as reliable indicators for AMD. The expected community response to both toxicity and inert solids is best described simply in terms of suppression of both taxon richness (*S*) and abundance (*n*) regardless of saprobity. A simple model (AMD′) is proposed that provides a precise and reliable metric of the effects of AMD in rivers.

1. Introduction

Acid mine drainage (AMD) is a common pollutant that is formed when pyritic rocks and ores are exposed to atmospheric oxygen either during the extraction of metal ores (e.g. Fe, Cu, Zn, Pb, As, U), sulphur or coal mining (Gray, 1998; Kim and Chon, 2001). In the presence of water and oxygen, bacterial mediated oxidation of the exposed rocks and minerals results in the rapid formation of a highly acidic and metal rich leachate, known as acid mine drainage, which can seriously impact both surface and ground waters (Singer and Strumm, 1970; Kelly, 1988; Evangelou and Zhang, 1995; Cherry et al., 2001; Saria et al., 2006; Lin et al., 2007).

The effect of acid mine drainage on rivers is dependent on their buffering capacity and available dilution (Kelly, 1988; Gray, 1997). However, expected impacts include a reduction in pH, elevated metal concentrations (e.g. Fe, Zn, Cu, Al, Pb, As, Cd, Mn, Se, etc.), the formation of ochre which is a stable orange precipitate comprising iron oxyhydroxides, and increased sulphate concentration (Gray, 1996). Thus the effects of AMD on rivers can be summarized as acidity, metal toxicity, metal precipitation and salinization (Gray, 1997). The relationship between AMD and the macroinvertebrate community of rivers has been widely studied (Thorpe and Lake,1973; Matter and Ney, 1981; Roline, 1988; Gower et al., 1995; Malmqvist and Hoffsten, 1999; Battaglia et al., 2005) with the level of impact reported ranging from non-detectable to complete destruction of the normal flora and fauna (Kelly, 1988; Gray, 1997; Cherry et al., 2001; De Nicola and Stapleton, 2002; David, 2003).

It is standard practice to assess the impact of pollutants on river ecosystems by the use of diversity and biotic indices that interpret changes in macroinvertebrate community (Hellawell, 1986; Rosenberg and Resh, 1993; Hering et al., 2004). Not surprisingly this has also led to their widespread use in the assessment of AMD in rivers, but in practice this has proven difficult and not always successful (Armitage, 1980; Chadwick and Canton, 1984; Whiting et al., 1994; Nelson and Roline, 1996). Biotic indices are highly specialized metrics, being used for a particular type of water pollution, normally organic pollution (e.g. Biological Monitoring Working Party index, EPT). In contrast, diversity indices are not specific to any particular type of pollutant but measure total environmental stress (e.g. Menhinick, Shannon, Brillouin indices) (Washington, 1984; Hellawell, 1986). Taxon richness can also be used as a measure of diversity but is susceptible to sample size, which is overcome by employing diversity indices that incorporate both taxon richness and abundance. Diversity indices are categorized as either dominance indices that are weighted towards abundance of the commonest species (e.g. Simpsons index) or information-statistic indices which are based on the rationale that diversity in a natural system can be measured in a way that is similar to the way information contained in a code or message is measured and so reflect taxon abundance (e.g. Shannon index, and Brillouin index) (Washington, 1984).

In this study the response of the macroinvertebrate community to AMD in the River Avoca, a poorly buffered erosional river impacted by AMD from an abandoned Cu–S mine in southeast Ireland, is assessed using both botic and diversity indices.

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Specifically the aim is to determine the reliability of using biotic indices for the assessment of AMD and to find a simple model of community response.

2. Methods

2.1. Location

The abandoned Cu–S mines at Avoca, County Wicklow, southeast Ireland, has been subject to on-going study since its closure in 1982 (Gray, 1998; Gaynor and Gray, 2004). Details of this river and the mines are given elsewhere (Sullivan et al., 1995). The acid mine drainage discharged from the site has seriously affected the water and biological status of the Avoca River, which is a highly erosional river forming the lower main channel of the Avonmore-Avoca Catchment (Watershed: 625 km^2; discharge rate at Avoca: 0.7–70 $m^3\,s^{-1}$), although there has been a steady recovery in quality as the acid mine drainage slowly alters in character (Gaynor and Gray, 2004).

The mines discharge into the Avoca River just downstream of the White Bridge (Ordnance Survey Map Reference T204768). The sample locations are shown in Fig. 1. Site 1 is the non-impacted control site 0.75 km upstream of the White Bridge and the mine adit discharges. Sites 2–5 all show visual signs of impaction by AMD in the form of orche deposition. Site 2 is located immediately after complete mixing at 2.5 km below the White Bridge and site 3 at 3.6 km. The River Aughrim is a major tributary that enters the river 7.25 km downstream of the White Bridge and which has an almost identical discharge rate to that of the main channel. Two more impacted sites are monitored below the confluence after complete mixing at site 4 at 8.5 km and site 5 at 11.0 km downstream of the White Bridge. The river enters the Irish Sea at Arklow 4 km blow site 5. Full details of the chemical and physical nature of the river have been discussed elsewhere (Gray, 1998; Gaynor and Gray, 2004).

2.2. Chemical analysis

Water and biological samples were taken at monthly intervals during the periods of lowest discharge rate from June to August in 2006, the only times when the river is reliably accessible for biological monitoring due to its variable and rapid changes in discharge rate with rainfall. Water samples were filtered as collected, in the field, through a Millipore cellulose nitrate membrane with a pore size of 0.45 μm, and stored in high-density plastic bottles and transported back to the laboratory for analysis in an icebox. Two sub-samples were taken, one being acidified for subsequent metal analysis, the other for sulphate, alkalinity, conductivity and pH analysis. Samples were stored at the laboratory in the dark at 4 °C. Conductivity, alkalinity and pH analysis were carried out within 24 h of sample collection using a WTW LF196 conductivity meter, the Gran titration method and a Jenway 3030 pH meter, with an Ag/AgCl reference electrode and temperature compensation, respectively (APHA, 1992). Sulphate

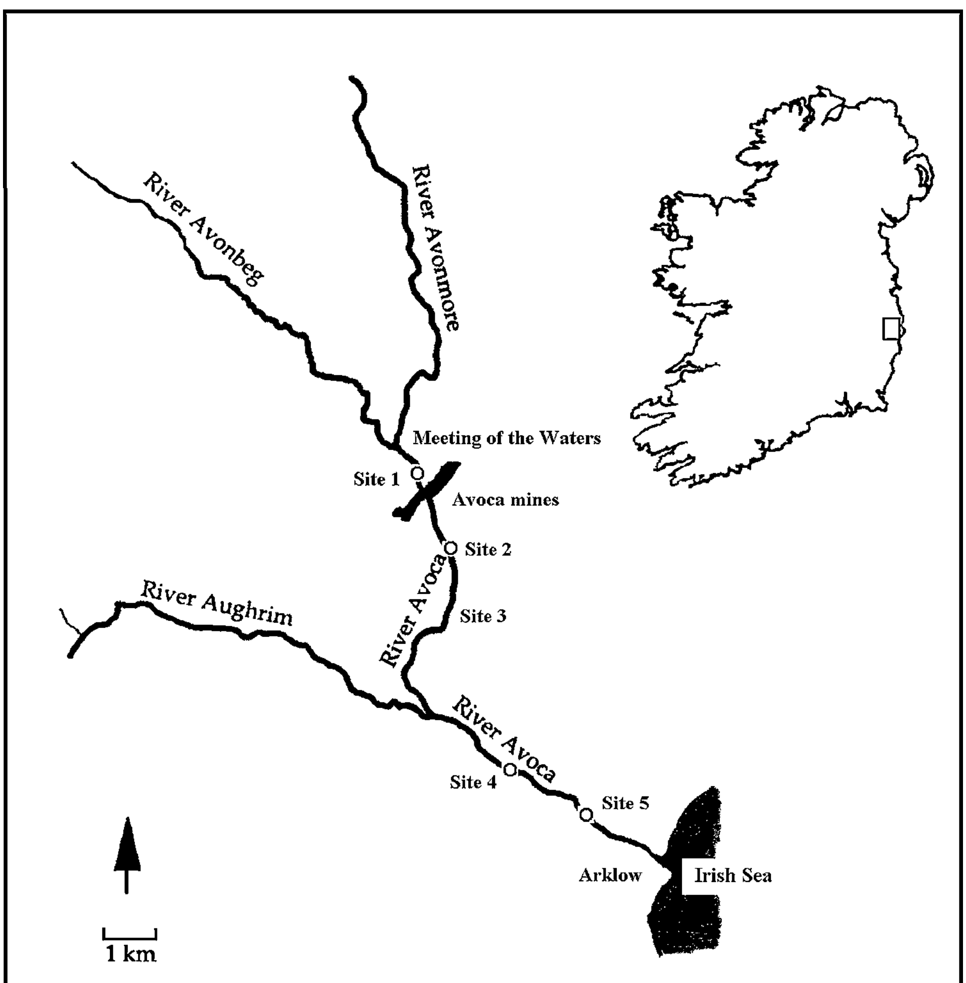

Fig. 1. Sample sites for biological and physico-chemical analysis along the Avoca River.

analysis was carried out by ion-exchange chromatography using a Dionex ICS-1500 analyzer. Cu, Fe, Zn, Pb and Cd were analyzed by flame atomic absorption spectrophotometry using a Perkin Elmer 3110 Atomic Absorption spectrophotometer with a graphite furnace (APHA, 1992).

2.3. Biological analysis

Macroinvertebrates were collected from riffle sites using a standard hand net (900 μm) collection technique (ISO, 1994). Collections were made for a total of 3 min using the replicate sampling technique of Byrne and Gray (1995) where multiple 30 s collections are made in an approximate transect across the width of the river at riffle or glide sites. Samples from each site were pooled and after preliminary sorting were stored in 75% alcohol and subsequently fully sorted in the laboratory with the animals identified to species level, except for the Chironomidae and Oligochaeta.

2.4. Indices

Five biotic and five diversity indices were selected based on their previous use for AMD assessment in freshwaters. The biotic indices were BMWP, ASPT, EPT, EPT/EPT + C and percentage Chironomidae. The diversity indices selected were the indices of Shannon, Simpson, Menhenick, Margalef and Brillouin.

The Biological Monitoring Working Party (BMWP) biotic score index was devised for surveillance of rivers in the UK, but has been widely adopted within Europe in modified form. Scoring is based on 85 families of known tolerance to organic pollution. The presence of each family provides a single score out of 10 (1 being highly tolerant, 10 highly sensitive) with the cumulative scores being the BMWP index. Due to the variation between scores due to sampling variation and seasonality, then it has become common practice to use the average score per taxa (ASPT) providing a single score out of 10 (Armitage et al., 1983; Hawkes, 1997).

$$\mathrm{ASPT} = \frac{\mathrm{BMWP}}{N} \tag{1}$$

where N is the total number of families used in the calculation of the total score (i.e. the BMWP). The EPT is the percentage abundance of the three important indicator orders, the Ephemeroptera (E), Plecoptera (P) and Trichoptera (T) (Hellawell, 1986).

$$\mathrm{EPT} = \left(\frac{\text{abundance of EPT}}{\text{total abundance}}\right)100 \tag{2}$$

The EPT uses the three most sensitive groups to organic pollution, however, the Chironomidae (C) are particularly tolerant to organic and other types of pollution including heavy metals. Therefore it can be included in the biotic EPT index to strength its ability to measure tolerance to pollution (i.e. EPT/EPT + C).

$$\frac{\mathrm{EPT}}{\mathrm{EPT}+\mathrm{C}} = \left(\frac{\text{abundance of EPT}}{\text{abundance of EPT} + \mathrm{C}}\right) \tag{3}$$

The final biotic index was the percentage abundance of Chironomidae. The diversity indices were calculated as (Pielou, 1966, (4); Simpson, 1949, (5); Menhinick, 1964, (6); Margalef, 1958, (7); Brillouin, 1962, (8))

$$\text{Shannon index}: \quad H' = -\sum_{i=1}^{S} \frac{n_i}{n} \log_e \left[\frac{n_i}{n}\right] \tag{4}$$

$$\text{Simpson's index}: \quad D = \sum_{i=1}^{S} \frac{n_i(n_i-1)}{n(n-1)} \tag{5}$$

$$\text{Menhinick's index}: \quad I = \frac{S}{\sqrt{n}} \tag{6}$$

$$\text{Margalef's index}: \quad I' = \frac{S-1}{\ln n} \tag{7}$$

$$\text{Brillouin's index}: \quad H = \frac{[\ln(n!) - \sum \ln(ni!)]}{n} \tag{8}$$

where S is the number of species in either a sample or population; n the total number of individuals in a population or community; n_i the number of individuals in the ith species.

A new index is proposed to specifically model the combined effects of both inert solids and toxicity. The AMD′ uses the mean of the number of taxon (S) and the total abundance (n), which it normalizes by taking the square root, per site:

$$\mathrm{AMD}' = \frac{S + \sqrt{n}}{2} \tag{9}$$

2.5. Statistical analysis

Differences between sites were based on the taxonomic composition of the macroinvertebrate assemblages measured using one-way non-parametric multivariate analysis of variance employing the Bray Curtis similarity index. Differences were also measured using macroinvertebrate abundance and taxon richness using one-way ANOVA. The relationships between the biological data, in terms of abundance and taxon richness and transformed using the various indices under investigation, and the chemical parameters were determined using the correlation of Pearson (De Pauw and Roels, 1988). This approach has successfully been used in similar studies previously (Camargo and García de Jalón, 1995; Zamora-Muñoz et al., 1995; García-Criado et al., 1999). All the biological parameters showed normal distributions while chemical variables were log-transformed where necessary to make them approximate to normality (García-Criado et al., 1999).

3. Results

The taxon richness and abundance for the control site (site 1) and impacted sites (sites 2–5) are summarized in Table 1. The diversity and abundance of the control site (1) shows a degree of variation that is consistent with normal seasonal variation which is often associated with temperature (Giller and Malmqvist, 1998) and so a seasonal mean is often used. Comparing the biological data collected in 1991, 9 years after the mines had been abandoned and the AMD was allowed to enter the river in an uncontrolled manner; the impact on the biota as measured by taxon richness (S) was far more severe in 1991 than 15 years later in 2006 ($p < 0.001$), 25 years after mine abandonment. Although the upstream control site (site 1) remains similar ($p > 0.05$) in terms of diversity, there has been an increase in productivity in this impoverished acidic river that is reflected in an increase in overall abundance. In 2006 the most impacted area appears to be between sites 1 and 2 with recovery occurring from site 2 downstream with 67% of the taxa re-established before entry to the sea. In contrast, the intensity of the impact in 1991 was continuous for much of the rivers length reaching the worse conditions at site 3 with little improvement after the confluence with the more buffered Aughrim River, which caused the remaining iron to be precipitated at sites 3 and 4 (Gray, 1998).

Table 2 shows that taxon richness is most strongly associated with the key abiotic indicators of AMD, namely pH ($p < 0.001$), sulphate ($p < 0.001$), Fe ($p < 0.01$), and Zn ($p < 0.001$), as well as alkalinity ($p < 0.001$) and conductivity ($p < 0.05$). Total abundance

Table 1
The taxon richness, total abundance and the AMD′ per sample site with the monthly mean and standard deviation (SD). (a) Using the 2006 survey data and (b) the 1991 survey data.

Site/month	Taxa					Abundance					AMD′				
	June	July	August	Mean	SD	June	July	August	Mean	SD	June	July	August	Mean	SD
(a)															
1	27	27	25	26.3	0.9	200	898	861	653	320	20.6	28.5	27.2	25.4	3.47
2	8	5	8	7.0	1.4	38	5	38	27	16	7.1	3.6	7.1	5.9	1.63
3	11	9	11	10.3	0.9	32	41	85	53	23	8.3	7.7	10.1	8.7	1.02
4	17	21	18	18.7	1.7	48	268	142	153	90	12.0	18.7	12.2	14.3	3.11
5	20	16	17	17.7	1.2	157	384	438	326	122	16.3	17.8	19.0	17.7	1.10
(b)															
1	29	34	28	30.3	2.6	160	141	131	144	12	20.8	22.9	19.7	21.1	1.33
2	1	2	1	1.3	0.5	52	2	23	26	21	4.1	1.7	2.9	2.9	0.98
3	1	0	0	0.3	0.5	12	0	0	4.0	6	2.2	0	0	0.7	1.04
4	1	1	0	0.7	0.5	3	5	0	2.7	2.1	1.4	1.6	0	1.0	0.71
5	2	2	0	1.3	0.9	2	67	0	23	31	1.7	5.1	0	2.3	2.12

Table 2
Pearson correlations (two-tailed) (r) of metrics, including AMD′, against key abiotic AMD parameters using the 2006 data.

Metric	pH	Sulphate ($mg\,l^{-1}$)	Conductivity ($\mu S\,cm^{-1}$)	Alkalinity ($mg\,CaCO_3\,l^{-1}$)	Fe ($\mu g\,l^{-1}$)	Zn ($\mu g\,l^{-1}$)	Average, r
AMD′	0.927***	−0.701**	−0.591*	0.775***	−0.611*	−0.875***	0.747***
Total abundance	0.804***	−0.520*	−0.557*	0.534*	−0.533*	−0.724**	0.612*
Taxon richness	0.875***	−0.791***	−0.521*	0.894***	−0.670**	−0.940***	0.782***
Brillouin	0.835***	−0.762***	−0.440	0.808***	−0.702**	−0.826***	0.729**
Shannon	0.716**	−0.690**	−0.410	0.746***	−0.164	−0.752***	0.580*
Menhinick	−0.434	0.177	−0.233	−0.101	0.258	0.261	0.244
EPT	−0.280	0.013	−0.517*	−0.061	0.003	−0.076	0.158
BMWP	0.825***	−0.775***	−0.580*	−0.844***	−0.685**	−0.918***	0.771***
ASPT	−0.172	0.222	−0.366	−0.242	−0.280	−0.030	0.219

* The level of significance is $p < 0.05$.
** The level of significance is $p < 0.01$.
*** The level of significance is $p < 0.001$.

Table 3
The relative abundance of the key macroinvertebrate taxa represented as EPH (Ephemeroptera), PLC (Plectoptera), TRIC (Tricoptera cased), TRIL (Tricoptera uncased), DIP (Diptera), CHR (*Chironomidae*), COL (Coleoptera), OLG (Oligochaeta), ARH (Arhnchobdellida), CRU (Crustacea), ARA (Arachnida) and LEP (Lepidoptera). The percentage representation of each family within the community is expressed as absent (–) 0, present (P) >5, few (FE) 6–10, frequent (FR) 11–25, common (C) 26–50, abundant (A) 51–75, very abundant (V) 76–100.

Taxa	Site 1			Site 2			Site 3			Site 4			Site 5		
	June	July	August	June	July	August	June	July	August	June	July	August	June	July	August
EPH	C	A	FR	FE	C	FR	FE	FE	P	FR	FR	P	FR	C	FE
PLC	FE	P	FR	FR	FR	FR	C	FR	P	FR	FE	FE	FE	C	FE
TRIC	C	P	P	C	FR	P	FR	FE	P	FR	FE	P	FE	P	–
TRIL	FR	P	P	FR	FR	FE	–	P	P	FE	P	P	FE	P	P
DIP	P	P	P	–	–	P	–	FE	P	–	P	P	P	P	CP
CHR	–	P	FR	FR	–	A	FR	C	A	FR	FR	A	FR	FR	C
COL	FE	FE	C	FE	–	–	FE	–	FE	FR	C	FR	FR	FR	FE
OLG	–	P	P	–	–	–	FR	FE	FE	FR	FE	P	C	–	–
ARH	–	–	–	–	–	–	–	–	–	–	–	–	P	–	–
CRU	–	–	P	–	–	–	–	–	–	–	–	–	P	–	–
ARA	–	–	P	–	–	–	–	–	–	–	–	P	–	–	–
LEP	P	–	–	–	–	–	–	–	–	–	–	–	–	–	–

was also correlated with parameters but not as strongly as taxon richness: pH ($p < 0.001$), sulphate ($p < 0.05$), Fe ($p < 0.05$), and Zn ($p < 0.01$), as well as alkalinity ($p < 0.05$) and conductivity ($p < 0.05$). The relative abundance (%) of key macroinvertebrate groups is summarized in Table 3 using the system of Iliopoulou-Georgudaki et al. (2003), although it should be remembered that the total abundances at the most impacted sites (sites 2–3) are very low.

4. Discussion

River quality is normally described using biomonitors, especially through the use of macroinvertebrates (Johnson et al., 1993). Indices are widely used to explore and express macroinvertebrate assemblages and to quantify the impact of pollutants on community structure (Washington, 1984; Hellawell, 1986). The biota of rivers responds to different pollutants in a characteristic manner. For example, the response to organic matter (Fig. 2a), toxicity (Fig. 2b) and inert solids (Fig. 2c) are expressed in terms of characteristic changes in the relative taxon richness and total abundance (UNESCO, 1978). However, AMD is a multifactor pollutant with its impacts identified as acidity, toxicity, formation of inert particles (i.e. ochre) and salinization (Kelly, 1988; Gray, 1997). Each of these impacts causes different effects on the community structure than organic matter on which the action of traditional biotic indices are based (Washington, 1984; Hellawell, 1986). For that reason diversity indices are recommended for use when the source of environmental stress is unknown (Hellawell, 1986). An exception to this is the family richness of Ephemeroptera,

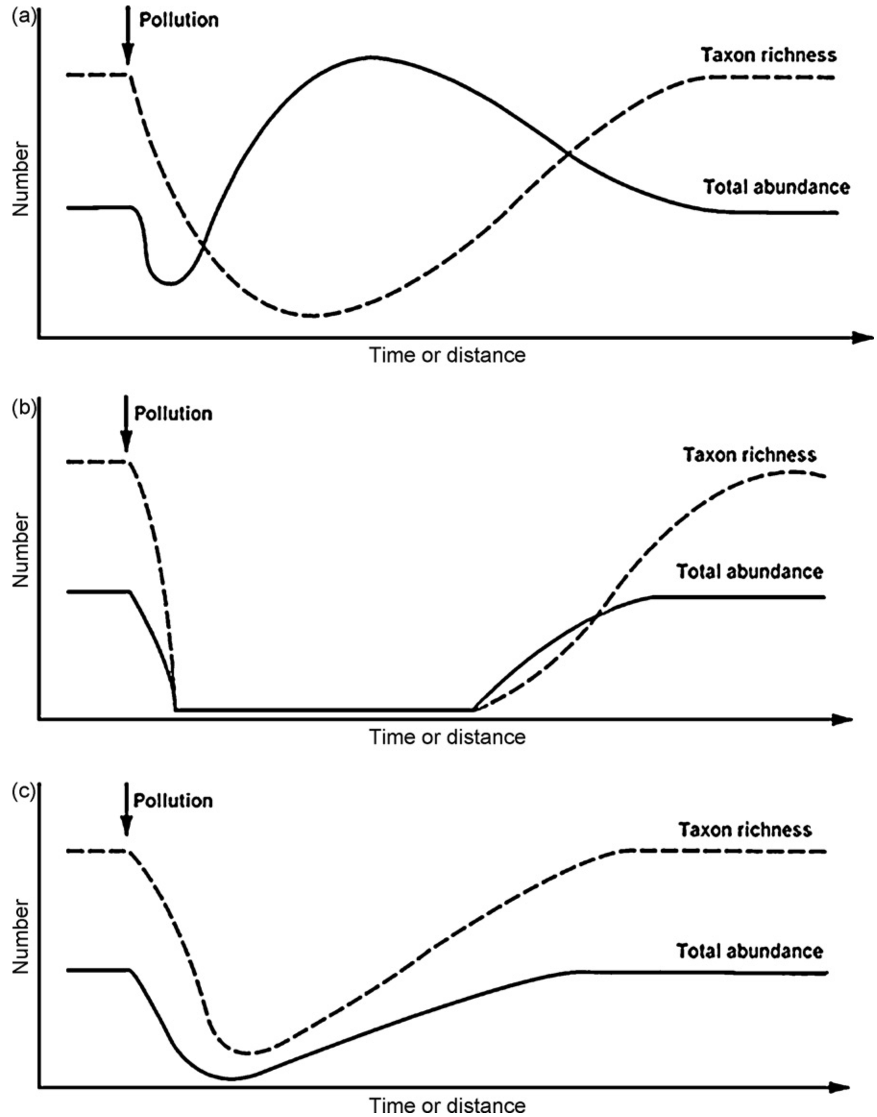

Fig. 2. Generalized model of the effect of organic non-toxic pollution on river macroinvertebrate community structure. Where (a) is the response to organic pollution, (b) to toxicity and (c) inert solids (UNESCO, 1978).

Plecoptera and Trichoptera (EPT), which has been found to be a useful indicator for acidity and also AMD as Ephemeroptera are sensitive to low pH and metals while Plecoptera are largely tolerant (García-Criado et al., 1999; Malmqvist and Hoffsten, 1999). However, this was not confirmed in the present study (Table 2).

Previous studies have indicated varying levels of success in employing indices in the assessment of AMD impact on benthic macroinvertebrate communities (Armitage, 1980; Chadwick and Canton, 1984; Whiting et al., 1994; Nelson and Roline, 1996), although the majority have recommended the use of the BMWP for routine surveillance at impacted sites. In a comparative study of the performance of indices in a river affected by coal mining in Spain and subject to acidic drainage, García-Criado et al. (1999) found the BMWP (modified for the Iberian Peninsula) and the EPT were the most highly correlated with mining impact. Similarly Malmqvist and Hoffsten (1999) reported reductions in taxon richness and EPT with AMD impact in Swedish streams, but not for total abundance nor specifically Trichoptera.

The results from this study show a significant reduction in both abundance and taxon richness in response to AMD. This is consistent with other similar studies who attributed this reduction to the loss of Ephemeroptera that are particularly sensitive to both pH and metals (Courtney and Clements, 1998; Battaglia et al., 2005), although Winterbourn and Collier (1987) who studied streams over a pH range of 3.5–8.1 in New Zealand found no correlation between pH and taxon richness. A particular problem with using EPT appears to be the seasonal variation of taxa from the three orders with Plecoptera declining with increasing temperature but Ephemeroptera and Tricoptera both increasing (Sporka et al., 2006).

When the community structure in a river impacted by AMD is considered then two levels of impact are normally discernable. When the impact is most severe (i.e. close to the discharge point, discharges are relatively acidic or metal rich, or the receiving water offers inadequate dilution), then the community response is similar to that recorded for toxicity (Fig. 2b), with all species eliminated, especially when the receiving waters have a low alkalinity and hence a poor acid neutralizing capacity. Downstream the most tolerant species (e.g. *Chironomidae*) return first but unlike toxic pollution rarely at elevated abundances. This is seen in the 1991 data set for the Avoca River (Table 1b), which mimics the response for toxic pollution. A similar response has been reported by Cherry et al. (2001). Both average taxon richness and abundance decrease with decreasing river pH (Giller and

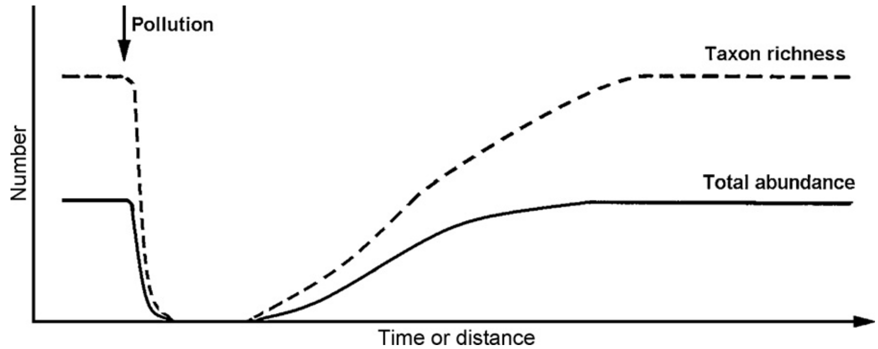

Fig. 3. Generalized model of the effect of a discharge of acid mine drainage on river macroinvertebrate community structure.

Malmqvist, 1998) resulting in a similar response to toxicity. Where the impact is reduced due to better buffering or greater dilution then the response is similar to that for inert solids (Fig. 2c), with a slow recovery in both taxon richness and abundance. The inert solids in this case are ferric oxyhydroxide particles that settle out of suspension coating the surface in the yellow-orange ochre that is characteristic of AMD impacted rivers (Gray, 1996). This response is described in Fig. 3. However, when the temperature is seasonally raised then the substrate becomes colonized by periphyton comprised primarily of iron bacteria, and occasionally tolerant filamentous green algae such as *Hormidium rivulare*, permitting greater taxon recovery by offering new niches and feeding opportunities. *H. rivulare* is tolerant to both heavy metals and acidification (Hargreaves and Whitton, 1976; Say and Whitton, 1977) and like the periphyton accumulates metals (Kelly and Whitton, 1989), potentially further restricting recolonization.

As expected from previous studies the BMWP showed a high degree of association with AMD parameters while ASPT did not. Of the diversity indices those based on information theory (i.e. the Brillouin and Shannon indices) performed best (Table 2). Although widely reported as being a good measure of the impact of AMD in rivers, the BMWP score is invariably unrelated to the ASPT (García-Criado et al., 1999). The BMWP score can vary significantly due to seasonal and sampling variation, so while the score gives an indication of taxon richness the ASPT is generally used to measure water quality as an average saprobic rating out of 10 (Hellawell, 1986; Zamora-Muñoz et al., 1995). The ASPT is not correlated with any of the key AMD abiotic indicators, showing that the impact is independent of saprobity, which is confirmed by the poor correlations between the abiotic parameters and the EPT (Table 2). Therefore the association recorded between BMWP and AMD impact is simply a function of taxon richness against which it is also highly correlated ($p < 0.001$). This makes the BMWP theoretically unreliable as are other biotic scores used for the assessment of organic pollution, for use with AMD impacted sites. The Brillouin index and taxon richness are the most reliable metrics of macroinvertebrate community response to AMD in the Avoca River. Information indices are more responsive to rare species; an important factor in the AMD impacted rivers where both abundance and taxon diversity are severely affected. The Shannon index does not respond to abundance as long as the proportional abundance remains constant, a situation that occurs with recovery from inert pollution, which is not the case with the Brillouin index, which makes it more sensitive to likely AMD scenarios. However, in practice the Brillouin index is complex to calculate and so less popular than the Shannon index.

All diversity indices are based on the theoretical response of macroinvertebrate communities to increasing stress (Routledge, 1979). The index is the degree the observed community varies from the assumed model on which water quality is then based. For diversity indices this response is based largely on the concept of high taxon richness with each taxon present in small abundances in clean waters. However, as stress increases then sensitive taxon are increasingly excluded with taxon richness falling but with tolerant taxon found at high abundances. In AMD impacted rivers abundance, like species richness, declines with increasing AMD impact (Fig. 3 and Table 2). So for AMD impacted rivers taxon richness and abundance appear to be the best and simplest descriptors of community structure.

The adopted model (AMD′) uses the number of taxon per site (S) and the normalized total abundance per site (n) (Eq. (9)). These two values were combined and the mean taken in order to smooth out the response of the community both spatially and temporally, which is often not possible when other metrics are used or either S or n on their own. The inclusion of abundance also takes into account the probability of drift organisms being sampled within the impacted zone, whereas dependence solely on taxon richness tends towards an overestimation of water quality. In general, $S > \sqrt{n}$ at impacted sites unless suffering from severe toxicity, so the inclusion of abundance both improves resolution of the metric and its sensitivity (Fig. 4).

The AMD′ is correlated with all the key AMD indicator parameters, especially pH against which it is very strongly correlated ($p < 0.001$; r^2 0.86) and alkalinity ($p < 0.001$; r^2 0.60), with negative correlations with sulphate ($p < 0.01$; r^2 0.49), conductivity ($p < 0.05$; r^2 0.35), Fe ($p < 0.05$; r^2 0.37) and Zn ($p < 0.001$; r^2 0.77) (Table 2).

Many researchers including this study have reported a good correlation between the BMWP score and the impact of AMD on benthic macroinvertebrates. This close correlation appears to be

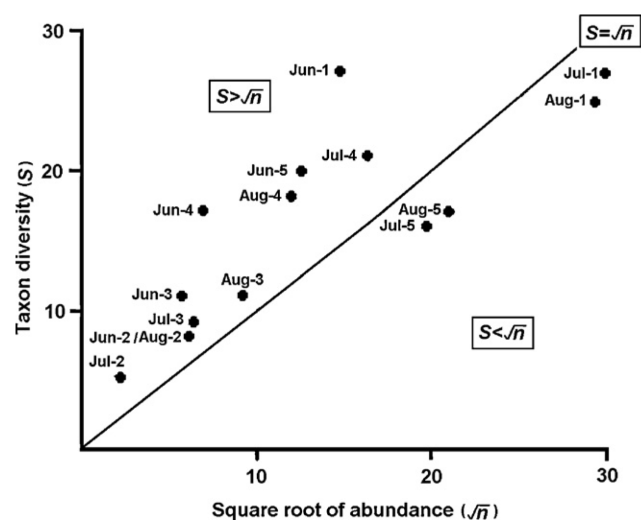

Fig. 4. A plot of taxon diversity against the square root of abundance for all sites (month-site) analyzed during the 2006 river survey of the Avoca River.

Table 4
Pearson correlations (two-tailed) (r) and level of significance (p) of widely used metrics against AMD′ using the 2006 data.

AMD′ vs. metric	r	p
EPT	−0.004	>0.05
EPTC	−0.201	>0.05
BMWP	0.855	<0.001
ASPT	−0.089	>0.05
Simpson	0.372	>0.05
Menhinick	−0.590	<0.05
Margalef	0.716	<0.01
Shannon	0.552	<0.05
Brillouin	0.725	<0.01
Chironomid (%)	−0.316	>0.05
Total abundance	0.914	<0.001
Taxon richness	0.914	<0.001

based solely on taxon richness and not saprobity as the ASPT, which is not influenced by sample size, seasonality or sampling effort, is unrelated to AMD concentration. A number of specific biotic metrics have been developed to assess acidification in surface waters (Raddum et al., 1988; Davy-Bowker et al., 2005) and also heavy metal pollution (Winner et al.,1980; Clements et al., 1992), which suggests that the development of a specific biotic index for AMD should be possible. However, the differing community responses to the varying components of AMD (i.e. inert (toxic) solids, metal toxicity, acidity and salinization) has prevented a specific biotic AMD index being developed.

While taxon richness is most strongly associated with AMD indicator parameters, the use of AMD′ smoothes out the variation in the taxon richness seen downstream in the recovery zone. By including normalized abundance then the AMD′ is able to model the responses for toxicity and inert solids more accurately than other indices. The relationship of the AMD′ with other indices is shown in Table 4.

5. Conclusions

1. AMD causes a different response to the riverine community structure, as measured by benthic macroinvertebrates, than that recorded for organic pollution and/or deoxygenation.
2. The impact of AMD is characterized as severe toxicity, acidification or inert solids depending on dilution and acid neutralizing capacity of the receiving water, the nature of the AMD and the distance from the input of the discharge.
3. In general the response of rivers to AMD is similar to that for toxicity where the impact is severe, or as for inert solids where the impact is less severe.
4. Biotic indices (e.g. BMWP) are based on saprobity, which does not accurately reflect community disturbance for either toxicity or inert solids.
5. Diversity indices measure total stress, and so are better at detecting AMD impact in rivers, but are based on a theoretical community model not wholly appropriate to AMD.
6. The simple model proposed describes the expected community response of both toxicity and inert solids and so is a precise and more reliable metric of AMD river impaction.

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