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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

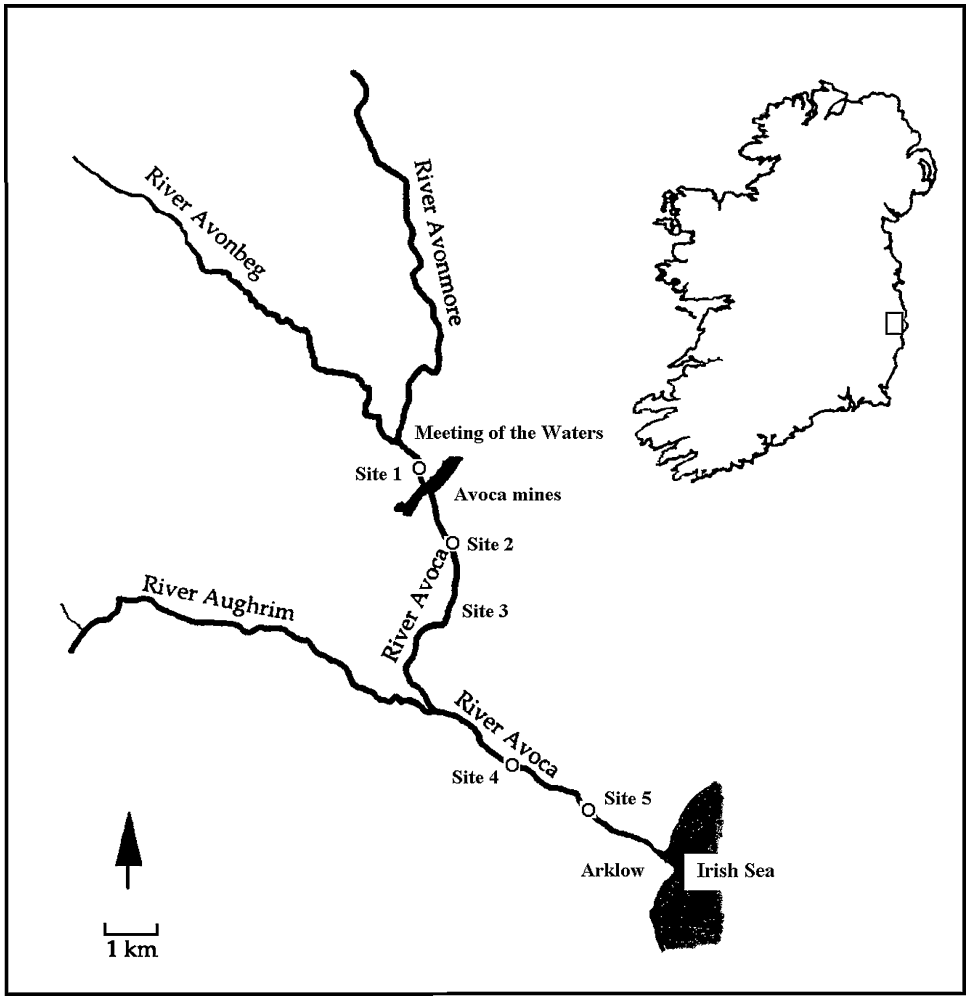

Fig. 1. Sample sites for biological and physico-chemical analysis along the Avoca River.