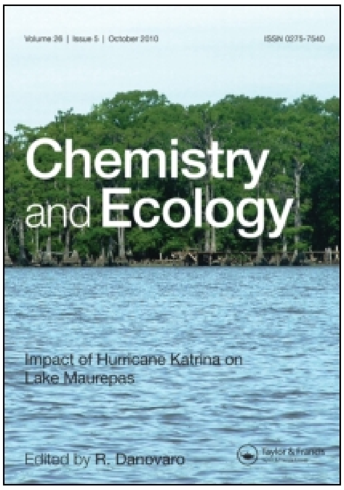

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Dominating Chemical Factors in Mine Water Induced Impoverishment of the Invertebrate Fauna of Two Streams in the Durham Coalfield, Uk

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DOMINATING CHEMICAL FACTORS IN MINE WATER INDUCED IMPOVERISHMENT OF THE INVERTEBRATE FAUNA OF TWO STREAMS IN THE DURHAM COALFIELD, UK

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Mine water pollution is causing increasing concern in the UK as some of the world's largest and longest-worked coalfields are abandoned and flooded. Many of the resulting polluting discharges are typified by higher mineral acidity (low pH), and high concentrations of iron and sulphate. Two streams, receiving mine water discharges in the Durham coalfield in northern England, have been studied to identify the major chemical factors affecting the welfare of benthic invertebrates. One of the discharges is strongly acidic (pH 3.9), and the second is marginally acidic (pH 5.3). Simultaneous analyses of hydrochemistry and invertebrate diversity and abundance demonstrate serious faunal impoverishment downstream of minewater discharges. Pathway analysis has been applied to ascertain statistically the dominating chemical factors in this faunal impoverishment. In both cases, total acidity (acidity to pH 8.3 = 612 mg l^{-1} as $CaCO_3$; pH 3.9) and iron concentration (up to 112.5 mg l^{-1}) account for nearly all of the faunal impoverishment. Other metals, sulphate and pH (in isolation from mineral acidity) are of lesser importance. Persistence of the detrimental effects of high acidity downstream of the discharges is ascribed to a slow recovery of the carbonate buffering system. These findings assist in the implementation of long-term management and remediation strategies for mine water pollution.

Keywords: Acidity; acid mine drainage; biodiversity; iron; mine water

INTRODUCTION

Mine water pollution is one of the most serious environmental impacts of mining, and affects at least 200 km of streams in England and

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Wales alone (National Rivers Authority, 1994). The phenomenon often occurs as a result of groundwater rebound, following mine abandonment. Groundwater rebound occurs when pumping activities cease, causing water table levels to rise. With the closure of Vane Tempest and Easington collieries, two of the last in the Durham Coalfield, United Kingdom (Fig. 1), the future of the long-term regional dewatering operations is uncertain. (Younger, 1993). Dewatering in this

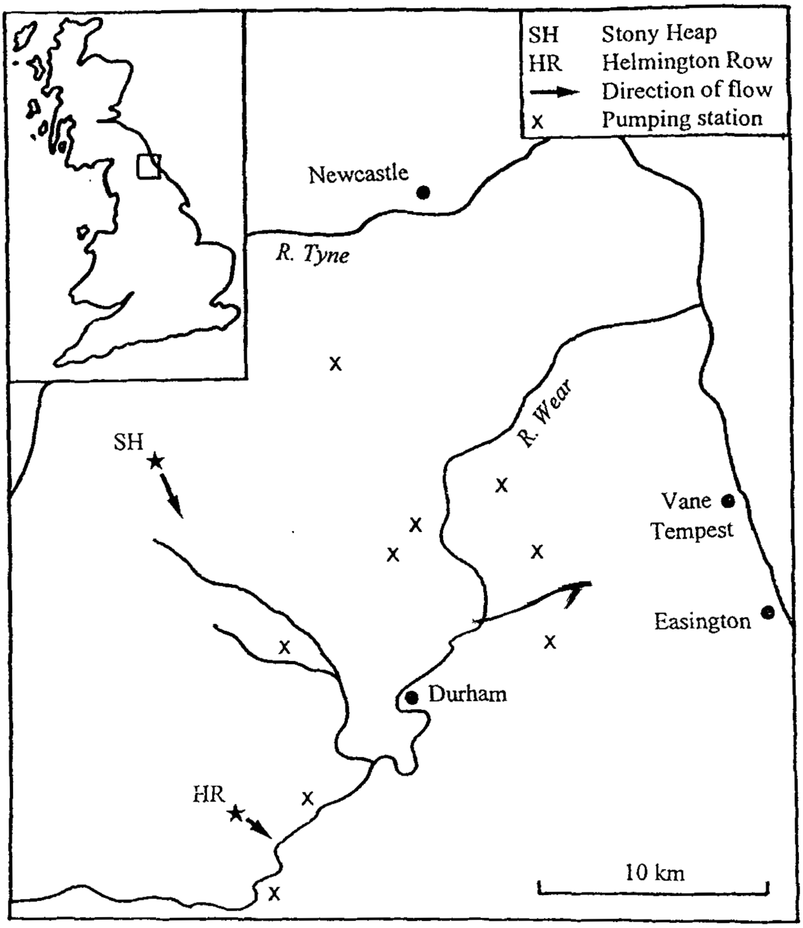


FIGURE 1 Location of study area.

context is defined as lowering of the water table through pumping activities, to enable access to deep coal seams. Contesting the original proposal of the state mining corporation (British Coal), to terminate dewatering activities at the abandoned collieries, are those who believe that the environmental repercussions of allowing groundwater rebound to proceed would be catastrophic for the aquatic environment of the River Wear. At the time of writing, dewatering operations continue, with the industry regulator (the Coal Authority) spending an estimated £1M per annum on pumping from nine shafts to maintain dewatering over an area of approximately 600 km^2 (Younger and Harbourne, 1995). However, the uncertainty regarding the long-term continuation of pumping has provoked considerable research into mine water pollution, since a cessation of dewatering activities may lead to a rapid increase in the number of mine water discharges across this 600 km^2 area. Particular tasks include predicting the timing and quality of future uncontrolled discharges following a cessation of pumping (Sherwood and Younger, 1994), and assessing impacts and planning remediation for streams already affected by mine water pollution (Younger, 1995). This paper outlines the current impacts of two mine waters, one that is strongly acidic (pH 3.9) and one that is circumneutral (pH 5.3), on the benthic macro-invertebrate fauna of streams in the Durham coalfield, and applies a pathway analysis to highlight the most important chemical factors. Such work has local significance as a contribution to the debate on the future environmental management of the Durham coalfield; in the international context, the insights gained have important implications for the "targeting" of key pollutants in the design of active and passive treatment systems for mine water remediation (cf. Hedin *et al.*, 1994).

CHEMISTRY OF MINE WATER POLLUTION

The mineral pyrite (iron disulphide, FeS_2) is widespread in the coal measures of the north-east of England (Younger, 1994). As water tables are lowered, through pumping activities, pyrite is exposed to oxidation. Subsequent groundwater rebound enables the oxidation residues to be washed into solution and, eventually, into surface streams. The chemical process can be summarised as follows:

$$4\,FeS_2 + 15\,O_2 + 14\,H_2O \rightarrow 4\,Fe(OH)_3 + 16\,H^+ + 8\,SO_4^{2-} \quad (1)$$

(From Barnes and Romberger, 1968).

The essential points, in the context of this study, are:

(i) oxidation of pyrite in mine workings may lead to the formation of highly acidic waters;
(ii) subsurface processes may neutralise much of the proton acidity (reflected in pH) in these waters; hence mine waters emerging at the earth's surface may be net-acidic (i.e. acidity > alkalinity) or net-alkaline (alkalinity > acidity) (Hedin *et al.*, 1994; Younger, 1995);
(iii) the mineral acidity which is endemic in net-acidic waters, and frequently persists in net-alkaline waters, leads to an increase in the concentration of metallic ions in receiving waters;
(iv) in the oxidising and/or neutralising environment of receiving waters, the high concentrations of iron are reduced by precipitation of orange or red iron hydroxides ("ochre") which frequently smother the stream beds and banks;
(v) sulphate concentrations persist typically at high levels in receiving waters, since there are few sinks for sulphate in an oxidising environment.

The chemistry of mine water is reviewed in much greater detail elsewhere (Barnes and Romberger, 1968; Singer and Stumm, 1970; Silver, 1988).

METHODS

Study Sites

The two systems under investigation are on the western border of the Durham coalfield (Fig. 1), but are beyond the influence of the current pumping regime (Younger, 1993).

The mine water discharge sites were initially identified by a geochemical survey of the entire region by Younger and Bradley (1994). Two particular discharges were chosen for this study as representatives of two contrasting styles of mine water pollution:

(i) The Stony Heap discharge (UK Grid Reference NZ 149518; S1 on Figure 2) arises from shallow underground workings. The discharge itself is marginally acidic, although it is often net-alkaline before and after the summer (i.e. alkalinity > acidity), despite having a pH which may be less than 6.5. The receiving watercourse, the Newhouse Burn, is net-alkaline throughout the year

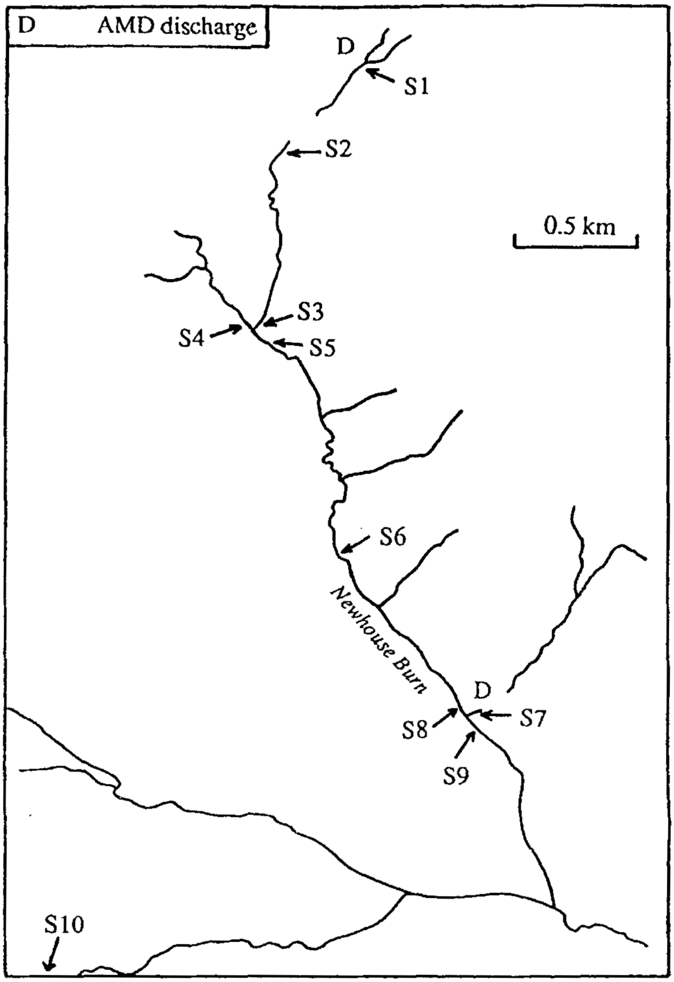


FIGURE 2 Sampling points along the Newhouse Burn, Stony Heap. (D = discharge)

and is part of the River Wear catchment, one of northern England's major river systems. The Stony Heap discharge (220 m a.s.l.) is some 5 km upstream of the Newhouse Burn's confluence with the River Wear. Several tributaries join it over this length, hence providing a substantial source of dilution. Above the mine water discharge it is no more than a drainage ditch, and is too small for invertebrate collection.

(ii) By contrast, the Helmington Row discharge (NZ 185357; H1 on Figure 3) appears to be colliery spoil leachate, and is always strongly net-acidic. The receiving watercourse is also acidic for several hundred metres downstream of the discharge. The affected stream is also part of the River Wear catchment, but it is only 2 km to the confluence, and there are far fewer diluting tributaries. The discharge emanates at 135 m a.s.l. Again, above the outfall there is only a very low flow in the channel.

The two steam systems receiving these discharges were sampled in June 1994 at a number of points downstream of these major

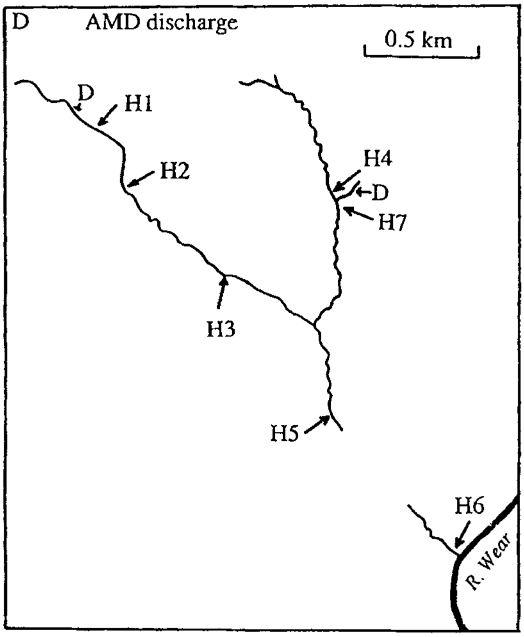


FIGURE 3 Sampling points along the Helmington Row stream. (D = discharge)

discharges (Figs. 2 and 3), so as to evaluate the spatial patterns of hydrochemistry and the faunal biodiversity. Sampling was undertaken during a period of dry weather. Precipitation generally leads to dilution of the impacts of the mine waters, due simply to increased flows in streams which are not matched by increased mine water flows.

Chemical Sampling and Analysis

At Stony Heap, the mine water discharge itself was sampled, together with nine points downstream of the discharge. One of these was a separate reference site, located on a nearby physically similar stream, which is not impacted by mine water. At Helmington Row the mine water discharge was sampled, together with six sites downstream of the mine water. Again, a separate reference site was used (H4 on Figure 3).

Temperature, pH and electrical conductivity were measured in the field using a Palintest digital probe. Temperature (°C) and pH were measured simultaneously. Electrical conductivity (EC) results are reported in $\mu S\ cm^{-1}$. The probes were washed with distilled water prior to each measurement, and were submerged at the sampling site until readings stabilised. Alkalinity and acidity were measured volumetrically in the laboratory, using N/50 H_2SO_4 (with methyl orange indicator) and N/50 NaOH (with phenolphthalein indicator) within 24 hours of sample collection. It is standard protocol to report results for alkalinity and acidity in mgl^{-1} as $CaCO_3$ (APHA, 1995). This method is accurate to within $\pm$ 5 mg/l. During storage, samples were kept at 4°C. Sulphate was measured by the gravimetric method with ignition of residue. Sub-samples were collected for metals analyses, which were immediately treated with 1 ml of concentrated nitric acid. Metal analyses were performed by Inductively Coupled Plasma AES (ICP-AES). Standard methods were followed throughout (APHA, 1995).

Invertebrate Collection and Analysis

Three minute kick samples were carried out at each site to collect benthic invertebrates (APHA, 1995). A standard pond net (a 250 mm by 230 mm frame, with a 1000 μm mesh net of depth 300 mm) was used for collection of samples. The standard procedure was followed in collection of invertebrates by this method. The operator faces downstream, with the pond net held in front vertically against the substratum. The stream bed was then vigorously disturbed by kicking for the standard three minute period (Mason, 1996). Since the objective of the investigation was essentially to compare the ecology of sites, the same physical stream characteristics were looked for at each site. In this case riffle zones were used throughout, mainly because they are shallower than pools, and therefore provide easier access. Samples were preserved in 70% ethanol with a few drops of glycerol. In the laboratory, samples were sieved (mesh opening 0.5 mm) to remove fine debris. After sorting, identification was aided by use of a stereoscopic microscope (magnification 15–75 *x*). Identification was performed predominantly to the family taxonomic level. The data collected were subsequently used to calculate the Biological Monitoring Working Party (BMWP) score for each site (Chesters, 1980). Designed originally to give a broad indication of the condition of rivers in the United Kingdom, the method requires that taxa are identified down to the family level. Families are assigned a score (between 1 and 10) according to their pollution tolerance. Thus, the pollution intolerant families such as the mayflies and stoneflies are given a score of 10. The individual scores are then summed to give the BMWP score (Mason, 1996). This biotic index is the one currently used by the UK Environment Agency for benthic macroinvertebrate analyses.

Pathway Analysis

Pathway analysis is a powerful form of multiple regression and allows correlation between independent variables to be investigated, as well as the direct cause-effect relationships between independent and dependent variables (Sokal and Rohlf, 1995). Thus, a clearer insight can be gained into the interactions between causes (iron concentration, acidity, sulphate concentration), the combination of which result in an effect (faunal impoverishment). For example, from the previous discussion of mine water chemistry it is clear that the acidity concentration and sulphate concentration in a mine water are inextricably linked, and it would therefore be unrealistic to view them as completely separate causes (or not) of faunal impoverishment.

In the calculations below a number of independent (or *predictor*) variables are regressed against a single dependent (or *criterion*) variable.

Different cause-effect relationships are hypothesised, including ones involving indirect relationships, as a powerful statistical means of ascertaining the most crucial chemical influences of a mine water discharge on the benthic macroinvertebrate ecology of receiving watercourses.

Whilst significance tests are applicable in multiple regression, the emphasis in a pathway analysis is in the strength and direction of relationships rather than the statistical significance of specific coefficients (Sokal and Rohlf, 1995). Significance tests are therefore not discussed in the following pages.

RESULTS

Chemical

Results of chemical analyses are presented in Tables I (Stony Heap) and II (Helmington Row). The outstanding features of the data are the pH values and iron concentrations at the first sampling stations on each of the streams. At Stony Heap a pH value of 5.3 was recorded at site S1. At site S2, 0.5 km downstream, pH was neutral. Total iron concentrations at sites S1 and S2 (29.04 mg l^{-1} and 10.86 mg l^{-1}) are considerably higher than all other iron concentrations along the course of the receiving stream (sites S3 to S9 on the Newhouse Burn), which vary around 1.0 mg l^{-1}.

At Helmington Row, pH readings of 3.9 at sites H1 and H2 are the lowest recorded. Values again rise abruptly further downstream, reaching a maximum of 8.3 at H5 (approximately 1.5 km downstream). An iron concentration of 112.5 mg l^{-1} at H2 is much higher than any other along the Helmington Row stream.

Other noteworthy points are very low dissolved oxygen concentrations at sites adjacent to the main mine water discharges at Stony Heap (0.5 mg l^{-1} at S1 and 4.5 mg l^{-1} at S7); this is related in part to consumption of oxygen during the oxidation of ferrous iron to ferric iron (Younger, 1997):

$$\mathrm{Fe}^{2+} + \tfrac{1}{4}\mathrm{O}_2 + \mathrm{H}^+ \rightarrow \mathrm{Fe}^{3+} + \tfrac{1}{2}\mathrm{H}_2\mathrm{O}. \qquad (2)$$

However, dissolved oxygen concentrations recover rapidly downstream of the mine water discharges (10.0 mg l^{-1} at S2 and 9.6 mg l^{-1} at S8).

TABLE I Chemical analyses of Stony Heap mine water discharge and receiving watercourse

Parameter	*Site*									
	S1	*S2*	*S3*	*S4*	*S5*	*S6*	*S7*	*S8*	*S9*	*S10**
Temperature(°C)	10.1	10.8	12.5	13.9	12.5	12.8	13.0	13.6	14.6	15.1
pH	5.3	7.06	7.59	7.30	7.20	8.00	7.08	7.92	8.13	7.80
Conductivity (μS cm^{-1})	295	816	795	1184	830	858	575	860	882	281
Dissolved Oxygen (mg l^{-1})	0.5	10.0	9.9	9.8	9.7	9.9	4.5	9.6	8.7	8.8
Alkalinity (mg l^{-1} as $CaCO_3$)	178	176	188	398	200	188	238	194	220	108
Acidity (mg l^{-1} as $CaCO_3$)	194	34	24	38	24	18	32	28	22	18
Sulphate (mg l^{-1})	324	253	312	418	255	238	114	223	77	54
Iron (mg l^{-1})	29.04	10.86	1.27	0.26	1.16	0.31	0.87	0.59	0.36	0.65
Aluminium (mg l^{-1})	0.25	0.26	–	–	–	0.34	–	–	–	0.20
Zinc (mg l^{-1})	0.05	0.04	–	–	–	19.70	–	–	–	< 0.01
Lead (mg l^{-1})	0.03	0.04	–	–	–	0.03	–	–	–	0.03
Copper (mg l^{-1})	0.04	0.04	–	–	–	0.07	–	–	–	0.02

*Reference site

TABLE II Chemical analyses of Helmington Row mine water discharge and receiving watercourse

Parameter	*Site*						
	H1	*H2*	*H3*	*H4**	*H5*	*H6*	*H7*
Temperature(°C)	16.2	13.1	11.3	12.4	14.1	13.5	13.7
pH	3.87	3.93	7.46	7.07	8.28	7.83	7.01
Conductivity (μS cm^{-1})	872	2030	832	442	792	1040	870
Dissolved Oxygen (mg l^{-1})	7.2	0.1	10.6	9.2	11.2	10.6	8.3
Alkalinity (mg l^{-1} as $CaCO_3$)	0.0	0.0	190	102	174	176	90
Acidity (mg l^{-1} as $CaCO_3$)	78	612	26	14	20	22	18
Sulphate (mg l^{-1})	389	1322	266	128	255	382	355
Iron (mg l^{-1})	–	112.5	–	1.05	–	0.13	1.07
Aluminium (mg l^{-1})	–	66.1	–	0.32	–	0.03	0.38
Zinc (mg l^{-1})	–	2.84	–	1.23	–	0.03	0.01
Lead (mg l^{-1})	–	0.13	–	0.02	–	0.02	0.02
Copper (mg l^{-1})	–	0.21	–	0.04	–	0.03	0.02

*Reference site

Sulphate values are high in the proximity of polluting discharges ($\leq$ 1320 mg l^{-1}), particularly H2, where extreme values occur for many variables (Table II). All of these results are comparable to those of the NRA (1996), who subsequently sampled the two discharges in September, October and November 1995.

Biological Data

The graphical representations of Figures 4 and 5 illustrate the diversity and abundance of invertebrates at each of the sampling sites. The predominance of the pollution-tolerant order *Diptera*, is clear (Mason, 1996). Also evident is the lower number of total individuals in samples from points of mine water discharge (that is S1, S2; H2, H7). The order *Ephemeroptera* are only found in any numbers at sites beyond the influence of the mine water discharges i.e. the upstream reference sites, H4 and S10. The information contained in these pie charts is reduced to a single numerical value for each site in Figure 6, where the values of the Biological Monitoring Working Party score are reported for Stony Heap. As expected the value for the reference site (S10) is higher than for all the others.

Pathway Analysis

In Figure 7 two pathways diagrams are shown. Selection of predictor variables was based on the results of Spearman's Rank correlations (Sokal and Rohlf, 1995), which indicated acidity, iron concentration, sulphate concentration and distance downstream to have the strongest relationships with the Biological Monitoring Working Party (BMWP) score. Single-headed arrows indicate a cause-effect relationship, the strength of which is given by the figure above the arrow, the pathway coefficient. It is apparent, therefore, that acidity and iron concentration are the predominant influences on the BMWP score, and hence the welfare of the biota. Notice that the pathway coefficients are negative, indicating that an increase in either acidity or iron concentration is reflected in a drop in the BMWP score. The direct influences of sulphate concentration and distance downstream are minimal. R^2 is the complete determination of the variance in Y(BMWP score) in terms of the three predictor variables. The high

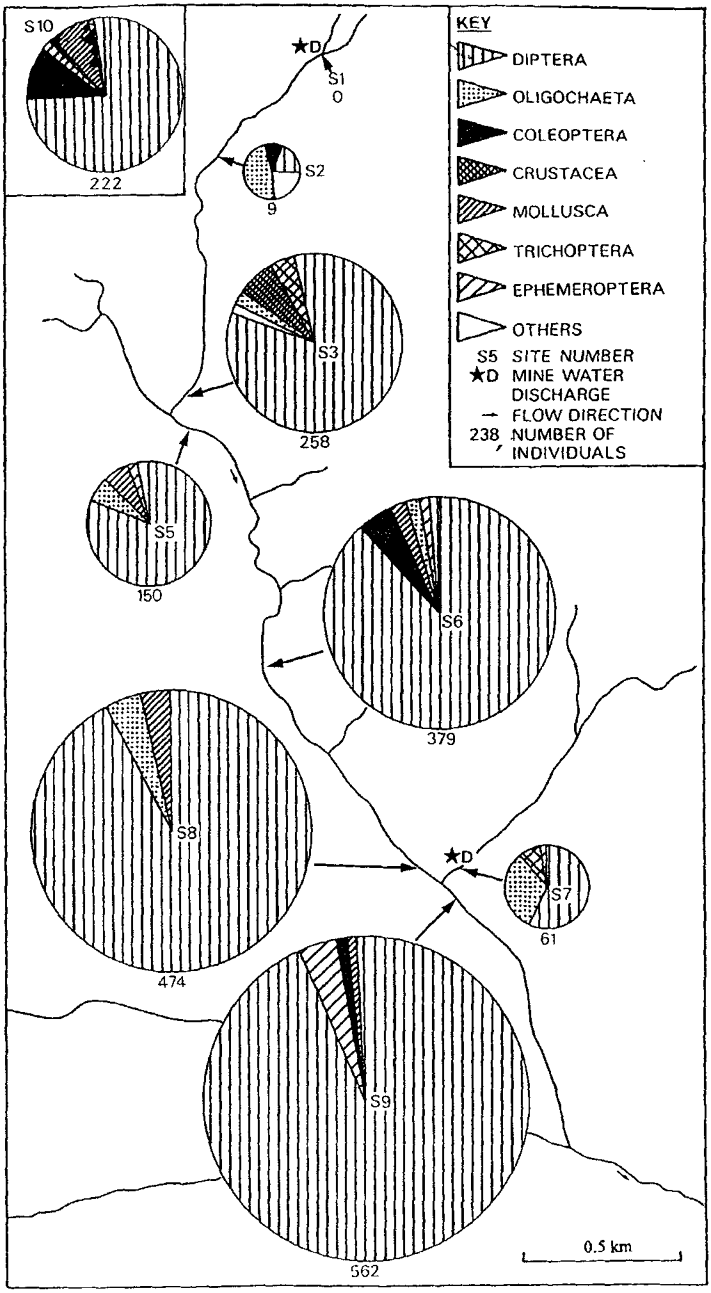


FIGURE 4 Graphical representation of invertebrate distribution along the Newhouse Burn, Stony Heap.

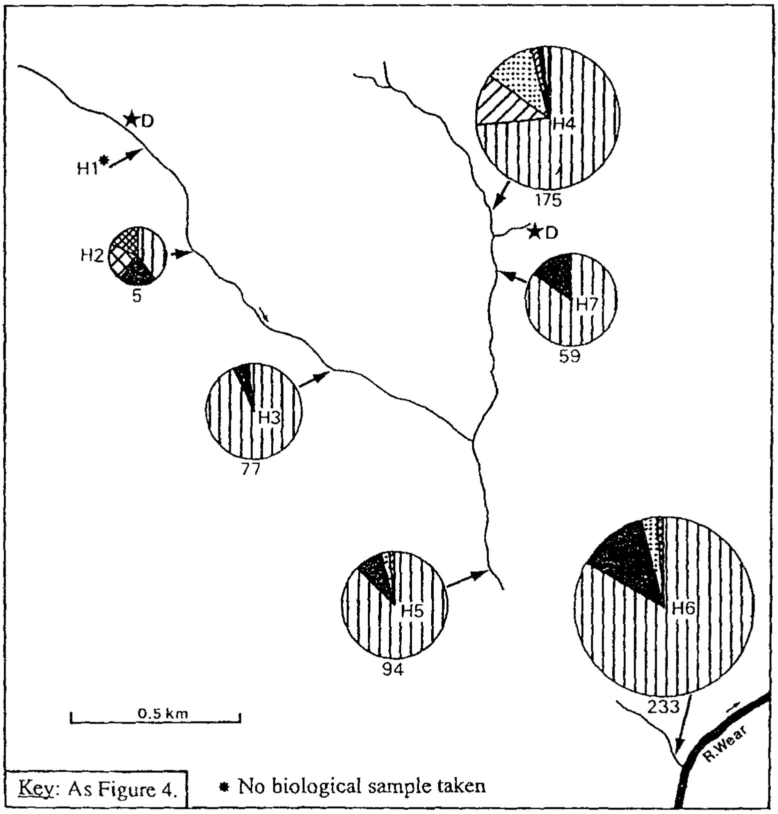


FIGURE 5 Graphical representation of invertebrate distribution along the Helmington Row stream.

values of 0.984 and 0.959 clearly implicate these three factors as the predominant influences on BMWP score. The respective values of the coefficient of multiple determination ($R^2_{Y.1,2,3}$), which are significantly lower than R^2 support the hypothesized indirect relationship of distance downstream and sulphate concentration.

DISCUSSION

Figures 4 and 5 illustrate both the distribution of groups and also the total number of individuals at each site. "Group" here is used to

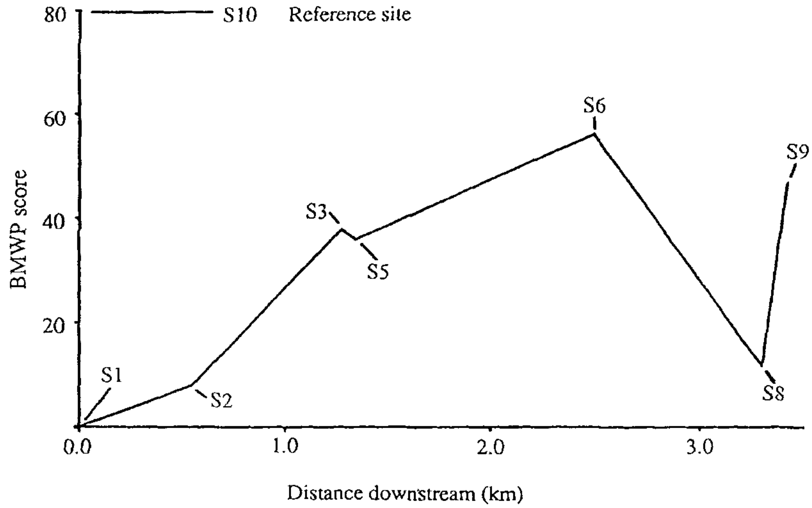
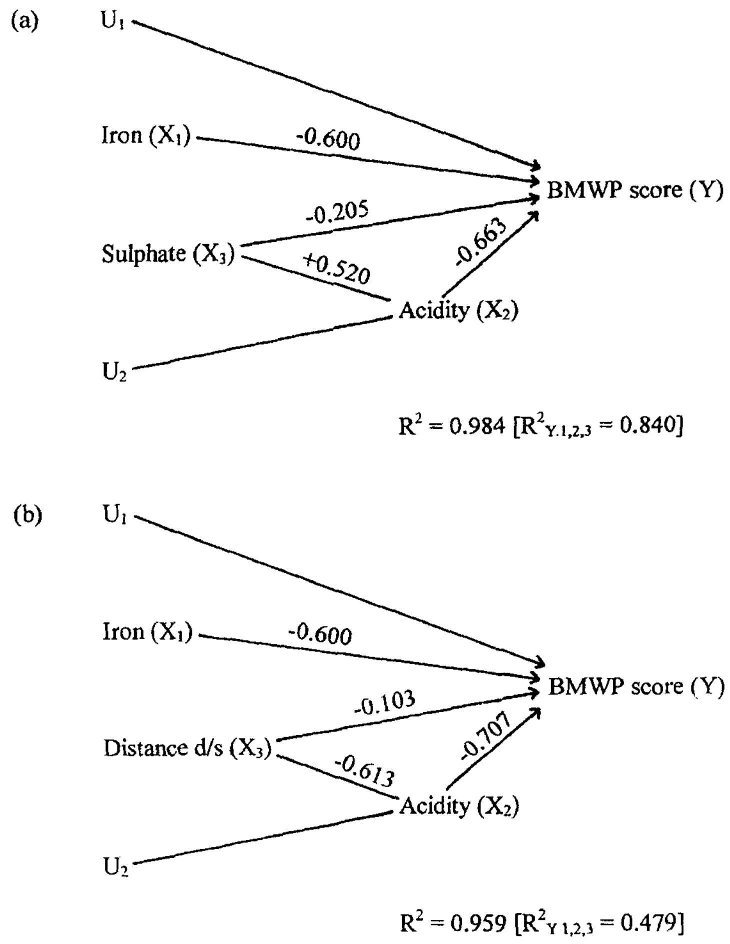


FIGURE 6 BMWP scores along the length of the Newhouse Burn, Stony Heap.

describe any category of invertebrates; divisions between categories are based on the ease of identification, and a group may therefore represent one family, or an entire order. The predominance of dipterans (which includes families such as *Chironomidae* and *Tipulidae*) is clear from an initial inspection of Figures 4 and 5. The Order *Diptera* constitute at least 75% of the total number of organisms at nine out of fourteen sites where samples were collected. Two of the exceptions are site S10, which is the reference site, and site H4, which is also beyond the influence of the polluting mine waters (and essentially a second reference site). Other sites where dipterans are not so prevalent are S2, S7 and H2-sites most acutely affected by mine water. This is simply because, with a very low total number of organisms, the contribution of a single invertebrate (as a percentage of the total) is greatly exaggerated. Also apparent is the recovery in the total number of organisms downstream of the point of first emergence.

However, 0 invertebrates at S1, 9 at S2, and 5 at H2, are clear indicators of the damaging effects of the mine water on the upper reaches of the two streams. Characterized by low pH and high iron concentrations, the implication is that these two parameters are instrumental in faunal impoverishment. The physiological stresses



$R^2 = 0.984$ $[R^2_{Y.1,2,3} = 0.840]$

$R^2 = 0.959$ $[R^2_{Y\,1,2,3} = 0.479]$

N.B. U_1 and U_2 are lumped parameters accounting for all of the variance in the criterion variable not described by the predictor variables.

FIGURE 7 Path diagrams illustrating primary causative influences on the BMWP score.

caused by low pH and high metal ion concentrations vary in their severity between invertebrates, but are similar in their *modus operandi*. Acidity disrupts the ionic balance across organism membranes (e.g. in gills, gut), and may cause denaturing of cell components. Calcium carbonate shells of molluscs and crustaceans may be dissolved (Kelly, 1988). In suspension, ferric hydroxide may limit light penetration, and more importantly in this case, as it precipitates out it can smother the benthos, acutely restricting invertebrate life. Inspection of the affected streams clearly illustrates this latter phenomenon to be a causal influence in faunal depletion, with thick deposits of ochre at sites S1, S2, S7 and H2.

Many of the observations of invertebrate distribution here are similar to those of other investigations of mine water impacts, particularly the dominance of dipterans. For example, in a study of the zinc-polluted Nent system of the northern Pennines of England, Armitage and Blackburn (1985) found an average of 38% of invertebrates to be Chironomids alone (one of the families within the Order *Diptera*), with dipterans providing 86% of all invertebrates collected in this study. Scullion and Edwards (1980) report a similar abundance of Chironomidae, as well as recovery downstream, but never to the levels of their reference site. Figure 6 illustrates this to be the case here as well. The reference site, S10, has a BMWP score of 80. Site S6 is ranked second, with a score of 56. It is therefore clear that whilst there is a gradual recovery downstream of the mine water discharge, the stream has an impoverished fauna along much of its length. Site S8, 3.3 km downstream of the point of first emergence, only has a BMWP score of 12 (see Figure 6). Whilst invertebrates are very abundant at this site (Fig. 4), the fauna is dominated by the low-scoring dipterans. The available chemical data shed no light on the reasons for this. However, there was evidence of ochre staining along the stream margins, and it may be that the mine water seepage into this reach of the stream had been more severe in the past (perhaps during the winter when water tables are high). Cairns *et al.* (1971) suggest that the rate of recolonization is a function of the distance from the source of the pollution and the in-flow of undamaged tributaries. This is apparent in this investigation, with good correlations between total number of individuals and distance downstream, and also greater numbers of individuals in the lower sites at Stony Heap than in comparable sites at Helmington Row, where there are fewer tributaries (see Figures 2 and 3). This latter point has important implications in view of any future proposal to terminate pumping activities in the centre of this coalfield. If this were to happen the estimated 105 $\mathrm{Ml\,d^{-1}}$ (Younger, 1993) of relatively clean water currently pumped into the catchment from the Coal Authority dewatering shafts (Fig. 1), which serves as a significant source of dilution, would no longer be available. The reduction in flow-rate would undoubtedly be to the detriment of the already acutely damaged biota, even before acid mine waters manifested themselves at the surface in greater volume.

Together, acidity and the sedimentation of ferric hydroxide certainly appear to be the main causal factors in faunal depletion. However, as the two often occur together it is difficult to determine the precise roles of each. Many researchers have encountered similar difficulties in separating the impacts of the major parameters (Scullion and Edwards, 1980). In an effort to clarify the roles of these critical factors, further scrutiny of the pathway analysis results (Fig. 7) is warranted. Iron concentration and acidity are used as two of the three predictor variables throughout, as it is reasonable to assume these factors to be major influences on invertebrate abundance and diversity. Sulphate concentration or distance downstream is used as the third variable, as we have already seen these variables to be important peripheral influences in biotic change. Using iron concentration, acidity and sulphate concentration as predictors of BMWP score (at Stony Heap), R^2 equals 0.984 for the pathway hypothesized (Fig. 7(a)). This compares well with a coefficient of multiple determination of 0.840, and suggests that the role of sulphate concentration in governing faunal richness is an indirect, one, through its association with acidity (see Equation 1). The path coefficient of 0.520 between X_2 (sulphate concentration) and X_3 (acidity), in Figure 7(a), indicates a positive relationship between the two variables. This may be interpreted either as a cause-effect relationship, or (perhaps more likely) as a simple common-origin relationship. The important point, however, is that the direct influence of sulphate levels on BMWP score is weak, reflected in a path coefficient (X_3 to Y) of -0.205. Figure 7(b) demonstrates distance downstream (that is, distance from the pollution input) to have a similar indirect relationship, but the direction of the cause-effect phenomenon is the opposite to that of sulphate concentration. Increasing distance from the source is mirrored by a reduction in total acidity, but the direct influence of distance on the BMWP score is minimal. Distance is a primary influence on many of the chemical parameters here only through its relationship with discharge and time. It is obvious that with all other factors held constant, distance will be of no detriment or advantage to the welfare of the biota.

In summary, the dominant influences of acidity and iron concentration are confirmed. In general the results show acidity to have a slightly greater impact on the biota (indicated by higher path coefficients) than iron concentration. However, it seems likely that at grossly polluted sites, where there are heavy deposits of ochre, the impact of elevated iron concentration may be equally, if not more, important. It is possible that the impacts of acidity may persist further downstream than the effects of iron concentration, despite the recovery in pH values. The most detrimental effect of high iron concentration is the heavy deposition of ochre, but this is only evident in reaches of the two streams in close proximity to mine water discharges. In contrast, even where pH values have recovered to neutrality the buffering system of the waters, which is vital to the survival of many organisms, may not be available. This is supported by reports that in some mine water affected streams the bicarbonate buffering system has taken months to recover from the effects of a slug discharge of acidic water. Parsons (1977) suggests that during continuous acidic flows aquatic organisms may develop some resistance to the deleterious impacts of the depleted buffering system. However, given the evidence of this study, many organisms appear not to be able to develop sufficient tolerance to recolonise affected watercourses.

CONCLUSIONS

1. Results of chemical analysis clearly indicate the two study streams to be affected by mine water pollution. The variables measured are comparable with previous data from the area, and also with those from other locations in Britain and elsewhere.
2. Pathway analysis demonstrates that the predominant influences on biodiversity and faunal abundance are the elevated acidity and the iron concentration (with the smothering effects of ferric hydroxide on the stream bed). Sulphate concentration has a minimal direct impact on these streams. Metallic ion concentrations (e.g. Zn, Cu), whilst high at specific sites (e.g. Zn concentration of 19.70 mg l^{-1} at site S6), seem to have little impact on the fauna in these streams, despite their known toxicity (Kelly, 1988). It seems probable that their impacts are exceeded by those of acidity and ferric hydroxide deposition.

3. For both the marginally net-acidic system at Stony Heap and the strongly net-acidic Helmington Row stream, iron concentration and total acidity are the principal agents of faunal impoverishment. This suggests that the distinction between the two mine water types, while an invaluable guide to treatment design (Hedin *et al.*, 1994), is not in itself a distinction between more-or-less polluting discharges also.
4. The close interrelationship between iron concentration and acidity complicates the objective of distinguishing their individual impacts. However, in the upper reaches of the two streams, adjacent to the mine water inputs, low pH and heavy deposits of ochre are considered to be jointly responsible for the acute faunal impoverishment. Further downstream invertebrate diversity and abundance recover to a degree, although the fauna is consistently characterized by a predominance of the pollution-tolerant Order *Diptera* (as found in other mine water-affected streams (Mason, 1996)). Total numbers of invertebrates increase downstream of the point of first emergence of the mine waters, but index values never attain the magnitudes recorded at the reference sites. This is ascribed to the inflow of other mine water-affected streams, and to the inability of the bicarbonate buffering system to rejuvenate itself.
5. The impacts of mine water ,pollution discharges on receiving streams persist for some distance downstream. Should the number of mine water pollution discharges proliferate (as would happen if pumping activities ceased) the impacts on the biology of the main channel of the River Wear might be very serious.
6. Remediation measures for mine water-affected streams should clearly focus on removing iron and acidity. Where net-alkaline mine waters are treated, the focus can be narrowed to iron removal. Since total acidity includes the mineral acidity ascribable to iron, appropriate treatment of net-acidic waters correctly takes acidity as its principal focus (Hedin *et al.*, 1994).

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