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INFLUENCE OF DRAINAGE FROM OLD MINE DEPOSITS ON BENTHIC MACROINVERTEBRATE COMMUNITIES IN CENTRAL SWEDISH STREAMS

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Abstract—We analysed the benthic macroinvertebrate species composition, taxonomic richness (as expected richness for 100 individuals), total abundance and biomass at 117 stream sites in the province of Dalarna. Partial least squares regression models were constructed from observations on undisturbed sites and used to predict these community parameters at sites exposed to elevated levels of copper, zinc, lead and cadmium resulting from leakage from old mine deposits. Species richness at undisturbed sites was positively related to the size of the catchment, pH, channel width, calcium concentration and the proportion of deciduous trees in the riparian zone. In streams with elevated metal concentrations, we found reductions in taxonomic richness for total macroinvertebrates, mayflies, stoneflies and combined EPT (Ephemeroptera, Plecoptera and Trichoptera), but not for that of Trichoptera nor total abundance or biomass. Copper and zinc were those metals showing strongest negative associations with richness. Some taxa, common at undisturbed sites, were missing at metal-polluted sites. These taxa were the mayflies *Ameletus inopinatus*, *Ephemerella aurivilli* and *Heptagenia dalecarlica*, the stonefly *Protonemura meyeri* and the caddisfly *Apatania* sp. © 1999 Elsevier Science Ltd. All rights reserved

Key words—benthic macroinvertebrates, copper, heavy metals, species richness, streams

INTRODUCTION

Metal pollution resulting from mining has a well-known negative effect on the biota and especially on populations of metal-sensitive groups such as crustaceans and mayflies (e.g. Hynes, 1960). When metal concentrations are high there are usually no problems in understanding the causes of effects on the biota but this might be harder when these effects are more subtle, e.g. when the concentrations are moderate and when there are mitigating effects of other substances such as calcium, humic substances and fine particulate organics (Gerhardt, 1993).

Human impact on global biodiversity is conceived as one of the major problems today (e.g. McNeely *et al.*, 1995). Most of this concern regards diversity in tropical rainforests and marine coral reefs, but recently it has been reported that freshwater systems, including running waters, in temperate regions also face serious threats, especially through habitat degradation (including pollution, damming and water diversion) and the accidental or deliberate introduction of nonindigenous species (Allan and Flecker, 1993).

In a study on species richness of macroinvertebrates and mosses in Swedish rivers, we developed a methodology to predict species richness and abundance at sites disturbed by hydroelectric schemes using models developed at sites not influenced by this disturbance (Englund and Malmqvist, 1996; Englund *et al.*, 1997a,b; Zhang *et al.*, 1998). In this paper, we report on a study of 117 stream sites in central Sweden using a similar technique. In this area there has been extensive mining for copper, iron, lead and zinc for centuries and the old deposits of mine-tailing and crushed rock are still causing a leakage of a mixture of heavy metals into the nearby streams. Concentrations are not very high, but this pollution has obviously been going on for a very long time. We tested whether such sites had a reduced species richness and if other community variables such as abundance and mass were affected.

An account highlighting the general features of the benthic stream fauna in this part of Sweden based on this material is reported elsewhere (Malmqvist and Hoffsten, submitted for publication). In the present paper focus is on the influence of mining.

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MATERIALS AND METHODS

Site description

The investigated sites are located in the province of Dalarna, which features forested lowlands over hilly forest terrain to mountainous heathland (Nordiska Ministerrrådet, 1984). Three quarters of Dalarna is covered by forest dominated by pine (*Pinus sylvestris*, 51%) and spruce (*Picea abies*, 40%) with some deciduous components (birches, *Betula pubescence* and *B. verrucosa*, 9%). A major biogeographic limit, known as "limes norrlandicus", touches the south-eastern part and here the terrain is rich in mires. The geology is characterised by gneisses and granites, except for the Siljan region, a relatively small area with lime-rich minerals and glacial moraine soils.

The climate is characterised by relatively warm summers and cold winters which are harsh in the north–west. The duration of the snow cover ranges between 125 and 175 d per year and annual runoff between 8 and 14 $l\,s^{-1}\,km^{-2}$ (Tryselius, 1971).

In the south-eastern parts the old mine deposits cause leakage of metals. Very high concentrations have been recorded in lake sediments (Bernes and Grundsten, 1991). Since heavy metals are more soluble in acid soils and waters their presence is related to pH.

The large gradients in environmental factors are obvious from the Appendix. In brief, human impact varies from relatively small to substantial. Sites are located at elevations between 82 and 736 m a.s.l. The catchments upstream of the sampling sites range between 1.5 and 2097 km^2 and the most common land use is forest (82%). Metal concentrations are generally low, but at polluted sites maximum levels of copper, zinc, lead and cadmium reached 34.8, 1,480, 16.5 and 3.4 $\mu g\,l^{-1}$, respectively.

Fourteen sites were assigned *a priori* as metal polluted, based on the presence of deposits of mine wastes, a fact that was verified through the chemical analyses (Table 1). Most of the sites are affected by copper mines (pyrite) or smelting houses some of which date back to the 16th century (Site numbers 25, 64, 65, 75), the 17th century (28, 35), 18th century (41, 86), 19th century (74, 81) and some are from this century (11, 20, 109). Site number 3 is affected by a steel industry since the 1950's. On average, at affected sites as compared with the unaffected sites, the levels of copper, zinc, lead and cadmium were 11, 118, 8, and 40 times higher, respectively. Copper concentrations were high or very high at these 14 sites according to the Swedish standards (Swedish EPA, 1990). Corresponding numbers of sites for zinc, lead and cadmium were 11, 5 and 7, respectively. Note that these are spot readings, whereas the standards relate to annual means of at least three years. Eight and nine sites, respectively, reached the limits for copper (6 μg/l) and zinc (100 μg/l), recommended in "Environmental Quality Standard" (Mance *et al*., 1984; Mance and Yates, 1984). Iron is included in Table 1 to show that concentrations of this metal were not elevated like those of the other metals at mine-affected sites.

Field methods

Animal and environmental samples were taken on a single occasion between the end of May and mid-June in 1991–1993. For animals, a Surber sampler (area 0.05 m^2, mesh 0.50 mm) was used at 117 sites distributed over the entire region (Fig. 1). Ten samples were taken in transects across the stream in riffles of similar quality throughout the study. In addition to animal samples, shading, instream and riparian vegetation, depth, velocity, width were measured and water samples collected for further analysis. Chemical analyses of total phosphorus and nitrogen, calcium, sulphate, copper, zinc, cadmium, lead, iron, pH, alkalinity, conductivity and absorbance at 420 nm, were performed by MeAna HB in Uppsala according to "Swedish Standard" on water samples brought back to the laboratory. Animals were separated from other material and identified to the lowest possible level. The wet mass of animal and ash-free dry mass of plant matter were also estimated at each site. The riparian tree vegetation was classified into species and relative cover (seven classes: 0 = absent to 6 = 100%), and later, to reduce the number of variables, we performed a principal component analysis on these data. The resulting scores for the first two components were used in the subsequent analyses to represent variation in tree vegetation near the sampling sites and called "forestPC1" and "forestPC2", respectively. In addition we created a variable giving the proportion on deciduous trees in the riparian zone.

Taxonomic richness

A number of problems make it a difficult task to estimate taxonomic richness at a stream site. Different species have different phenologies, i.e. the probability of finding a species varies with the season; some have short larval

Table 1. Metal concentrations in water samples at the 14 *a priori* classified metal-affected sites. Concentrations are presented in $\mu g l^{-1}$. Site numbers refer to the numbers in the map (Fig. 1). For comparison mean and standard errors of the unpolluted sites are given (N = 94)

Site	Copper	Zinc	Lead	Cadmium	Iron
3 Avesta	8.8	18	0.7	0.022	405
11 Brossån	2.8	6	0.1	0.016	249
20 Finnhytteån	0.9	100	1.0	0.146	29
25 Garpenbergsån	16.2	765	1.5	1.100	70
28 Gruvbäcken	6.0	780	16.5	3.360	288
35 Hosjön	1.0	3	0.2	0.004	195
41 Insjögruvbäcken	34.8	410	0.9	0.675	343
64 Magasinsbron a	9.2	465	0.7	0.130	296
65 Magasinsbron b	8.3	152	0.8	0.043	260
74 Pålsbenningån	3.4	6	0.4	0.008	430
75 Rullsån	3.8	300	7.9	1.040	853
81 Saxbäcken	5.9	1480	0.4	1.230	390
86 Stollbäcken	1.7	105	5.8	0.055	190
109 Ö. Klingen	6.0	95	0.4	0.070	60
Unpolluted sites					
Mean	0.6	2.8	0.4	0.014	521
Standard error	0.08	0.24	0.05	0.002	5.2

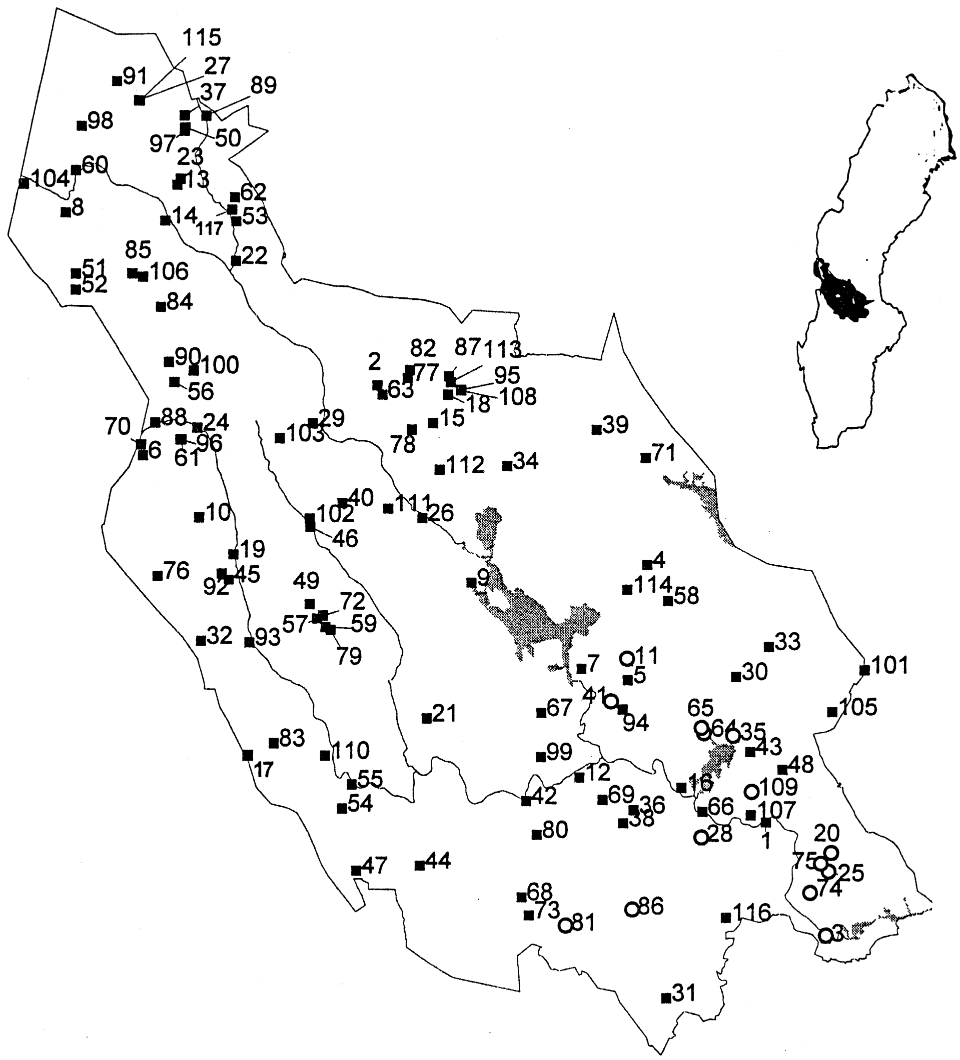

Fig. 1. Map of the investigation area showing the geographical locations of the study sites in the province of Dalarna in Central Sweden. Metal polluted sites are indicated by open circles. 1 Anstaån, 2 Aspvasslan, 3 Avesta, 4 Bjutjärnsbäcken, 5 Bjurtjärnsån, 6 Björnbäcken, 7 Bodaån, 8 Borgan, 9 Bossån, 10 Bredvallen, 11 Brossån, 12 Bröttjärnån, 13 Bunnan, 14 Byggningen, 15 Böån, 16 Dalbäcken, 17 Digerbergsbäcken, 18 Dyverån, 19 Feman, 20 Finnhytteån, 21 Fisklösbäcken, 22 Fjätälven, 23 Foskan, 24 Färdsjövallen, 25 Garpenbergsån, 26 Gopalån, 27 Grundöjen, 28 Gruvbäcken, 29 Gryvlan, 30 Gråsalfädbodbäcken, 31 Gärdsjöbobäcken, 32 Gärman, 33 Hafsbäcken, 34 Hedvasseln, 35 Hosjön inloppet, 36 Hyttingsån, 37 Hålbäcken, 38 Högbergsån, 39 Idån, 40 Ingelibäck, 41 Insjögruvbäcken, 42 Klarbäcken, 43 Knivaån, 44 Korplammsbäcken, 45 Kråkskibäcken, 46 Kumbelrodbäcken, 47 Kvarnbäcken, 48 Kvarnsjöbäcken, 49 Lejdbäcken, 50 Lill-Fjäten, 51 Lillån a, 52 Lillån b, 53 Lisselån, 54 Lissgranån a, 55 Lissgranån b, 56 Lisshöljan, 57 Lisslyån, 58 Ljusacksbäcken, 59 Lyån, 60 Långåbäcken, 61 Lägerdalsbäcken, 62 Lövhögsbäcken, 63 Lövåsbäcken, 64 Magasinsbron a, 65 Magasinsbron b, 66 Milsboån, 67 Mörkån, 68 Mörtån, 69 Noran inl., 70 Närån, 71 Nöttjärnsån, 72 Ogsjötjärnbäcken, 73 Pillisoån, 74 Pålsbenningån, 75 Rullsån, 76 Rullån, 77 Rymman, 78 Råvasseln, 79 Röbobäcken, 80 Salån, 81 Saxbäcken, 82 Skidbågsbäcken, 83 Skördrisån, 84 St. Göljån, 85 St. Njupån, 86 Stollbäcken, 87 Stopån, 88 Stordalsbäcken, 89 Storfjäten, 90 Storhöljan, 91 Storån, 92 Strandkölsbäcken, 93 Suskölsbäcken, 94 Svedjebäcken, 95 Svedvasseln, 96 Syndan, 97 Sågbäcken, 98 Sågslättbäcken, 99 Sångån, 100 Särkån, 101 Sörjabäcken, 102 Tennån a, 103 Tennån b, 104 Tollån, 105 Toxbäcken, 106 Trollvasslan, 107 Tvärhandsån, 108 Unnan, 109 utlopp Ö. Klingen, 110 Vallen, 111 Vasseln, 112 Våmån, 113 Västbysångsbäcken, 114 Årängsån, 115 Åstjärnsbäcken, 116 Åssån, 117 Öjvasseln.

stage, others long. Most aquatic insects have terrestrial adult stages, some have terrestrial pupae. In the present study, we assumed that most species were available as larvae in late May–early June, but it is clear that some taxa, e.g. winter stoneflies, could have been missed. However, it is not likely that this would cause any problems concerning the predictive models, because sampling time was compressed and there is no systematic difference between sites that were metal polluted and those that were not, with respect to species phenologies.

Another problem is related to the fact that rare species will be missed. There is a well-known relationship between the sampling effort and resulting number of species known as the "collector's curve" showing that the addition of species initially is rapid, but that the increment of species levels off as the number of samples increases. Species are also added at varying rates, e.g. where there is a high equitability (few species are very common) species richness increases rapidly with the number of individuals sampled, whereas at low equitability (some species very common, many species very rare) richness increases only slowly. Since species richness increases with the number of individuals collected, an error is also introduced if samples with greatly disparate number of individuals identified are compared. We accounted for this by using rarefaction analysis (Hurlbert, 1971; Krebs, 1989). This method standardises samples to numbers per individual and sample area, and we report the number of taxa for 100 individuals ("ET100", i.e. expected taxa richness for 100 individuals), because it was shown that this number was virtually as sensitive as using 300 individuals ("ET300", expected taxa richness for 300 individuals; Malmqvist and Hoffsten, submitted for publication, see also Vinson and Hawkins, 1996). The advantage of using the lower number is that more sites can be included in the analyses; samples from 33 sites produced fewer than 300 individuals, whereas only four had fewer than 100. At one of the polluted sites, Finnhytteån, the ET100 could not provide an observed value because the total abundance was only 92 individuals, which means that the sample size of metal-affected sites in richness-related analyses is based on 13 observations. Rarefied data were only used for total taxonomic richness.

Predictive models

Predictive models were used to forecast taxa richness, abundance and biomass of benthic macroinvertebrates at metal polluted sites using data from sites not having elevated metal levels (sites affected by organic pollution were likewise not used for model construction). We used SIMCA-S (version 6.0 for Windows) to build these models, using partial least squares analysis (PLS) on rarefied richness data. PLS reduces the number of variables to one or several latent components and the significance of these components was calculated in a cross-validation procedure (Martens and Næs, 1989). We used a large number of environmental variables (Appendix), excluding heavy metal concentrations, to create these primary predictive models, based on unpolluted sites only. These initial models indicated which variables contributed most to richness, abundance and mass observed. Significance was assessed through cross validation of each PLS model, where a Q^2-value for a significant model, or for a component, should be larger than a critical value ($Q^2_{\mathrm{limit}} = 0.097$ corresponding to $p < 0.05$, SIMCA Software Manual, 1996). Subsequently, the initial models were used for predicting the community properties for sites affected by metal pollution. The observed and predicted values were compared and the residuals provided a measure of the strength of the disturbance. If 95% confidence intervals of the means of this measure did not include zero, the effect was regarded as significant (Englund and Malmqvist, 1996). A new PLS analysis was performed using only the metal variables Cu, Zn, Cd and Pb at metal-polluted sites in order to separate the effects of the different metals. The procedure was repeated for total abundance and biomass.

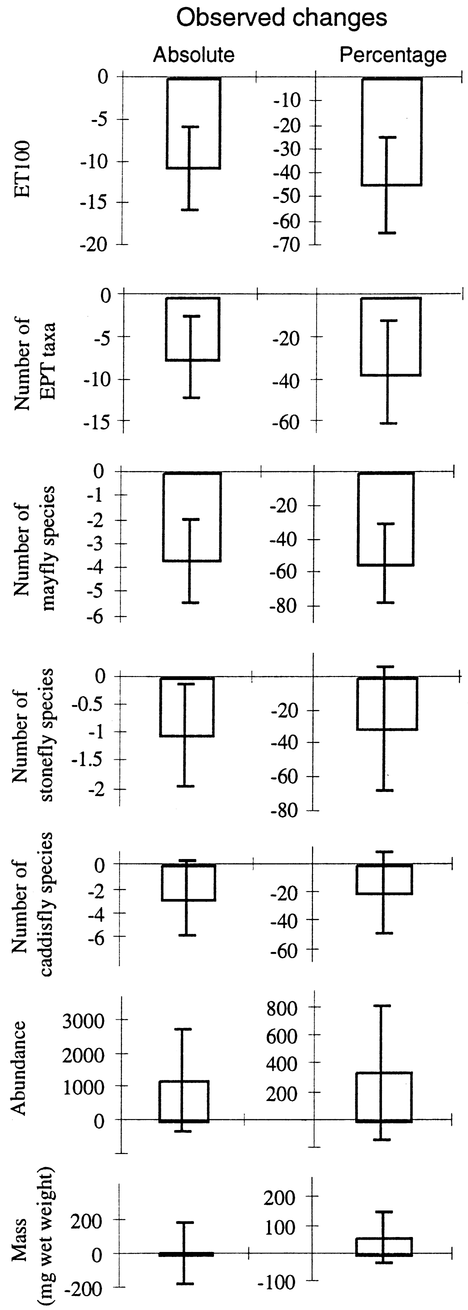

Fig. 2. Effect of metal pollution on taxonomic richness (ET 100), richness of mayflies, stoneflies and caddisflies, abundance and biomass. Negative values show average loss of taxa (as number of taxa and as percentage change), positive values average gain in abundance and mass (as numbers and mg wet weight, respectively, and as percentage change). 95% confidence limits excluding the zero line indicate significance.

Table 2. Summary of PLS models used to predict taxonomic richness, abundance and biomass at metal-affected sites. r_x^2 is the proportion of the variance in the environmental matrix used in the model, r_y^2 is the proportion of the variance in the dependent variable explained by the model and Q_v^2 is the proportion of the variance which can be predicted by the model (see the description of predictive models in the text for an explanation). The variables are ordered according to importance and the PLS scaled and centred regression coefficients are given in brackets. Only the five most important variables are shown. See Appendix for a complete list of environmental variables

	r_x^2	r_y^2	Q_v^2	Most important variables
Richness				
ET100	0.137	0.281	0.110	area (0.094), pH (0.086), width (0.083), calcium (0.070), deciduous (0.067)
Mayflies	0.135	0.561	0.460	area (0.122), pH (0.115), width (0.109), alkalinity (0.099), calcium (0.092)
Stoneflies	0.198	0.277	0.172	heath (0.060), altitude (0.059), forestPC2 (−0.059), iron (−0.057), velocity (0.054)
Caddisflies	0.129	0.397	0.427	algae (0.106), area (0.100), deciduous (0.095), pH (0.089), width (0.087)
EPT	0.137	0.502	0.427	pH (0.110), area (0.100), deciduous (0.099), width (0.094), calcium (0.088)
Abundance	0.140	0.268	0.117	latitude (−0.076), alkalinity (0.066), pH (0.066), algae (0.065), forestPC2 (−0.064)
Biomass	0.141	0.325	0.172	alkalinity (0.085), absorbance (−0.076), iron (−0.074), TOC (−0.071), farmland (0.067)

All data deviating from normality were log- or arcsine-transformed prior to analysis.

RESULTS

Metal-affected sites were characterised by reduced species diversity. The PLS models indicated that species richness (as ET100), on average, was 44% lower (or 11 taxa poorer) than predicted (Fig. 2). At affected sites, EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa richness was 36% lower than predicted. Looking at these taxa separately it is clear that taxa richness of mayflies was strongly reduced, stoneflies only marginally and caddisflies not at all (Fig. 2). There was no significant effect on either abundance or biomass (Fig. 2).

Taxonomic richness and EPT richness were strongly related to the size of the catchment along with pH and channel width, but also calcium, proportion of deciduous trees in the riparian zone, alkalinity and mass of benthic algae were important (Table 2). Mayfly richness was associated with similar variables. For stoneflies other variables, in particular proportion of mountain heath in catchment, elevation, current velocity and species composition of the riparian forest, were most strongly related to species richness. Iron and one forest composition variable showed negative relations. Taxonomic richness of caddisflies was favoured primarily by algal mass, catchment area, proportion of deciduous trees in the riparian zone, pH and channel width.

Abundance increased according to the PLS analyses with a more southern location, high alkalinity

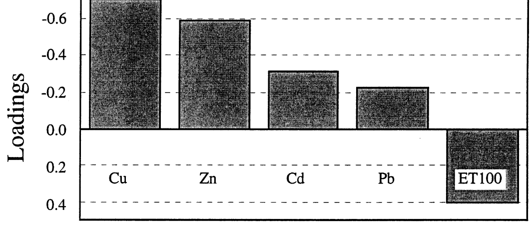

Fig. 3. Relative influence by the various metals on taxonomic richness (ET100) as estimated in PLS regression shown as the standardised coefficients (loadings).

and pH and high benthic algal mass (Table 2). Riparian vegetation was also of importance. Biomass appeared to be favoured by high alkalinity but also by farmland land use and low levels of colour, iron and TOC. Excluding one site (#3, Avesta), polluted by a steel industry rather than mining did not significantly change any of these results.

A separate PLS analysis showed that copper and zinc were stronger negative variables than lead and cadmium (Fig. 3). The model was significant ($r_x^2 = 0.562$, $r_y^2 = 0.346$, $Q_v^2 = 0.135$). A linear regression showed that richness decreased with copper concentration at sites *a priori* classified as metal affected (Fig. 4; $R^2 = 0.361$; $p = 0.030$).

Five taxa were recorded at ≥25% of nonaffected sites, but were absent from affected sites. These taxa were the mayflies *Ameletus inopinatus*, *Ephemerella aurivilli*, and *Heptagenia dalecarlica*, the stonefly *Protonemura meyeri* and the caddisfly *Apatania* sp.

Mayflies were completely lacking at the site with the highest zinc concentration (site #81 Saxbäcken, 1480 $\mu g\ l^{-1}$). Gruvbäcken (site #28; 780 $\mu g\ l^{-1}$) also lacked mayflies, but at this site lead and cadmium levels were high too (16.5 and 3.4 $\mu g\ l^{-1}$, respectively).

DISCUSSION

Studies of factors governing macroinvertebrate distributions have shown that scale is important (e.g. Allan and Johnson, 1997). In our study, the taxonomic richness at unpolluted sites showed clear relationships with regional factors, primarily catchment size, altitude and water quality, largely reflecting regional geology, but also local factors, including riparian vegetation, algae and channel width. Water quality in terms of pH tends to have a strong impact on macroinvertebrate community structure when sites include markedly acid streams (Hildrew and Giller, 1994) and this variable indeed proved to be positively associated with richness and abundance in our study. In contrast, iron appeared to be a factor negative for stonefly richness and biomass. Many streams in Dalarna are naturally acidic

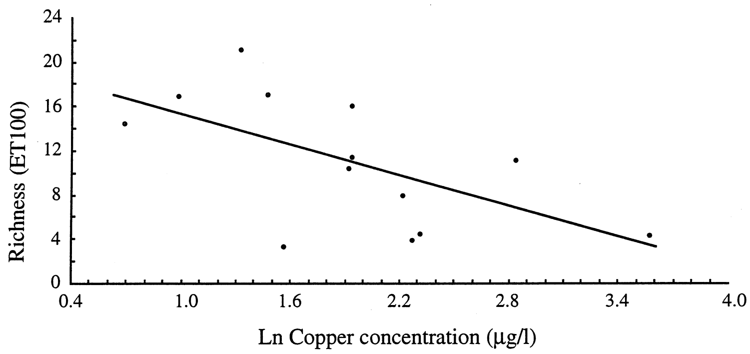

Fig. 4. Richness as a function of copper concentration at sites *a priori* classified as metal affected.

and humic and show relatively high concentrations of iron. Interestingly, iron concentrations were unrelated to concentrations of copper, zinc, cadmium and lead, which is also why this metal is not highlighted in our study. In other studies, iron deriving from mine fields has been shown to still cause problems decades after abandonment (Vuori, 1995).

Streams in this region of Sweden are apparently affected by metal pollution despite fairly low concentrations. Relatively few taxa, however, seem to be excluded and the effects are primarily seen as a reduced number of taxa per site. Reductions in taxonomic richness were seen especially in mayflies and stoneflies, and some taxa appeared to be especially sensitive, including *E. aurivilli*, *H. dalecarlica*, *P. meyeri* and *Apatania* sp. One common mayfly species, *A. inopinatus*, was absent from the sites with elevated metal levels. However, we can not conclude that it is metal sensitive, because it seems systematically restricted to relatively high-altitude streams. Several other studies have also reported on reduced richness following heavy metal pollution. Thus, Clements and Kiffney (1995) observed reduced overall richness and abundance, number of mayflies and abundances of mayflies and stoneflies in heavy metal polluted Rocky Mountain streams. They observed stronger effects at a higher elevation and on taxa with small body size (Kiffney and Clements, 1996).

In contrast to some published studies, we could not observe decreases in abundance nor mass. Of course, the metal levels in our investigation were comparatively modest, although the effects on richness and some taxa were evident. We think these effects might have been caused by higher concentrations in the substratum and/or in food material, such as detritus and algae, than in the water column. Also, concentrations at other times than when the spot readings were made might have shown higher values.

Our analysis of the relative effects of heavy metals on taxonomic richness suggested copper and zinc to be those with the strongest negative effects. However, this result should be regarded with caution, because of the correlative approach. Copper has well documented negative effects on stream animals. Thus, Leland *et al.* (1986) reported on changes in the benthic macroinvertebrate communities at copper concentrations as low as 5 $\mu g\,l^{-1}$ and the effect was greater on herbivores and detritivores than on predators, grazing mayflies being affected most severely. The authors argued that the number of taxa and similarity indices were better indicators than abundances, mass and measures of diversity, which to some extent is supported by our observations. Changes in community structure in streams claimed to be caused by copper have been observed, e.g. in England (Gower *et al.*, 1994) and the U.S.A. (Nimmo *et al.*, 1996), and strongly negative effects by this metal have been reported on the mayfly *Ephemerella dorothea* (Schultheis *et al.*, 1997), which suggests, when taking our observation on *E. aurivilli* into consideration, that members of this genus may be particularly sensitive. Besser *et al.* (1998) found that copper and cadmium, to a greater extent than zinc and lead, were accumulated by invertebrates and fish in a Colorado river affected by abandoned mines. Concentrations in grazers (the heptageniid mayfly *Rhithrogena*) were greater than those in predators (the stonefly *Megarcys*) and omnivores (the caddis *Arctopsyche*).

Our data indicated that zinc, along with copper, might be responsible for the reductions in taxonomic richness observed. This metal is reported to have significant effects on stream biota. E.g. Armitage (1980) found that zinc negatively affected taxonomic richness and abundance of macroinvertebrates in an English river system at concentrations between 300 and 2000 $\mu g\,l^{-1}$. In his study, some taxa appeared tolerant, especially the stoneflies *Amphinemura sulcicollis* and *Leuctra inermis*, and orthocladiine midges. Nine out of 13 mayfly species were restricted to sites with concentrations below 300 $\mu g\,l^{-1}$. The tolerance limit for *Baetis rhodani*, *Amphinemura sulcicollis*, *Isoperla grammatica*, *Leuctra fusca* and *Oreodytes* sp. was 1080 $\mu g\,l^{-1}$, a concentration exceeded only at a single site in our

Table 3.

	Min	Max	Median
Regional variables			
Coordinates from the Swedish National grid:			
x	13101	15316	14084
y	66521	68918	67624
Altitude (m a.s.l.)	81.9	736.1	350.6
Slope (%) 550 m	0.00	10.49	1.17
Catchment area (km^2)	1.54	2097.82	17.68
Bedrock inclination of weathering (0–1)	0.01	0.63	0.23
Land use (%)			
Lake	0.00	14.96	0.65
Forest	6.17	100.00	82.29
Wetlands	0.00	67.04	3.58
Mountain heath	0.00	78.10	0.00
Farmland	0.00	53.48	0.00
Urban	0.00	87.40	0.00
Water chemistry			
Cu ($\mu g\ l^{-1}$)	0.10	34.80	0.60
Zn ($\mu g\ l^{-1}$)	0.30	1480.00	2.55
Pb ($\mu g\ l^{-1}$)	0.04	16.50	0.21
Cd ($\mu g\ l^{-1}$)	0.0015	3.36	0.009
Fe ($\mu g\ l^{-1}$)	7	2980	340
Mg ($mg\ l^{-1}$)	0.11	21.8	0.49
Ca ($mg\ l^{-1}$)	0.52	244	2.6
SO4 ($mekv\ l^{-1}$)	0.015	13.4	0.041
pH	4.50	7.63	6.44
Conductivity ($mS\ m^{-1}$)	0.84	12.6	2.48
Alkalinity $mmol\ l^{-1}$	0	1.857	0.103
TOC $mg\ l^{-1}$	0.4	18.3	6.7
Tot – N $\mu g\ l^{-1}$	50	2750	260
Tot – P $\mu g\ l^{-1}$	2	157	8
Absorbance 420 nm	0.012	0.461	0.119
Local variables			
Canopy openness	1	5	3
Clearfelling (0 none, 1 one side 2 two sides)	0	2	0
Width (m)	0.7	50	3
Depth (m)	0.1	0.5	0.2
Current velocity (m/s)	0.21	0.91	0.60
Substrate			
Boulders	0	4	2
Cobbles	0	6	4
Pebbles	0	5	3
Gravel	0	4	2
Sand	0	5	2
Silt	0	5	0
Twigs	0	3	1
Riparian trees			
Alnus incana	0	5	2
Betula verrucosa/pubescence	0	4	3
B. nana	0	3	0
Salix sp.	0	4	2
Prunus padus	0	3	0
Acer platanoides	0	2	0
Sorbus aucuparia	0	2	0
Populus tremula	0	2	0
Picea abies	0	5	2
Pinus silvestris	0	4	1
Juniperus communis	0	5	0
Vegetation in water (mg ash – free dry mass)			
Batrachospermum	0	862.2	0
Other benthic algae	0	186.5	8.5
Fontinalis	0	2201	0
Other mosses	0	1212	8.2
Total vegetation	0	2856.7	128.2
Benthic organic matter (mg ash – free dry mass)			
FPOM	0	317	0
CPOM	0	508.2	91.9

study. Thorp and Lake (1973) reported that both the number of species and abundances were negatively impacted by zinc and cadmium in a Tasmanian stream exposed to mine wastes. In their study decreases in crustaceans, molluscs and worms were most obvious.

Where heavy metal pollution exists, it is not an easy task to tell which of the metals has the greatest effects on the animals since they often occur in high concentrations simultaneously. In the present study, copper and zinc showed the strongest negative correlation with taxonomic richness (copper was also the only metal that showed a significant negative relationship in bivariate correlation) and therefore, these metals might be the most important ones, which was also indicated in the above comparison with published data. Possible synergies between the metals and their effects on aquatic organisms are, however, not well known (Gerhardt, 1993).

The best studied aquatic insects with respect to ecotoxicology are probably chironomids. The metals studied here (Cu, Zn, Cd, Pb) have been shown to reduce growth and increase mortality of larval chironomids in laboratory experiments, especially that of cadmium on low-instar larvae of *Chironomus riparius* (Timmermans *et al.*, 1992). Adsorption on insect body cuticle was found to be low, corroborating findings by Seidman *et al.* (1986), and dead larvae had similar uptake rates as living larvae, indicating that the genuine uptake was low. If this holds true for other aquatic insects, passive uptake could be important. However, results of laboratory experiments can not easily be extrapolated to insects in nature (Hare, 1992) and uptake via contaminated food must be considered a potential route in metal-polluted streams. Moreover, sediments act as sinks for pollutants and many metals, like lead, are hydrophobic and therefore tend to be associated with particles (Vermeulen, 1995).

Most studies obviously show a reduction in the number of taxa following metal pollution. Reduced abundances and biomass of macroinvertebrates can be expected to affect such processes as grazing pressure on algal populations and energy transfer to higher trophic levels. A recent study demonstrated that the breakdown rate of leaf detritus was reduced in an American stream system with elevated copper concentrations (12–32 $\mu g\,l^{-1}$; Schultheis *et al.*, 1997). Obviously, heavy metal pollution can alter macroinvertebrate abundance and biomass, however, it is not at all clear how ecosystem processes are influenced by reductions of taxonomic diversity. We can only speculate that if "keystone" taxa, i.e. taxa with strong effects on the communities, are affected it could lead to much stronger impact than if functionally "redundant" taxa suffer the most. The problem is that we have very limited information from lotic systems about which taxa can be classified as keystone taxa.

CONCLUSIONS

1. The PLS prediction approach showed high sensitivity in detecting disturbances caused by metal pollution and should provide a useful instrument in determining effects of any environmental disturbances.
2. Our observations confirm the results of other studies that have shown reduced taxonomic richness as a consequence of metal pollution. In contrast, we found no effects concerning abundances or biomass.
3. This study unveiled particular reaction patterns to heavy metals in members of the macroinvertebrate fauna of small to medium-sized streams in Central Sweden. The most negative effects were recorded for mayfly species richness, whereas the richness of stonefly larvae was only marginally affected and that of caddis larvae remained unaffected.

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APPENDIX

Min-, max- and median values of environmental variables used in the PLS models are shown in Table 3.

Substrate variables are subjectively classified from complete cover (6) to absence (0). Riparian tree species are classified in the same manner. Similarly, canopy openness ranges from complete at 6 to deep shade at 0.