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Seasonal and Species-Dependent Variability in the Biological Impact of Mine Wastes in an Alpine River

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Biological impact assessments are now a common feature of many major industrial projects in North America. Although such studies should provide environmental data which contribute to the best overall use of watershed resources, they are often limited in scope and consequently may not account for natural biotic variability. While the diverse, changing nature of aquatic communities in unpolluted waters is well documented there is surprisingly little acknowledgement of such complexities in many impact assessments. The purpose of this investigation was therefore to show that the effects of mine wastes on aquatic communities depend on many factors, including 1) species under investigation, 2) season, 3) ability of species to adapt to pollutants.

MATERIALS AND METHODS

Collections of benthic invertebrates, algae, and sediments and water for chemical analysis were made adjacent to a tungsten mine during 1975 in the Flat River (61° 58' N, 128° 14' W) in a mountainous region of the Canadian Northwest Territories. The river, which is 200 km long, originates at an elevation of 2200 m, and flows to the South Nahanni River and eventually to the Mackenzie River. Samples were taken near the headwaters, where the river is 3 - 10 m wide, 10 - 30 cm deep and flows at 10 to 50 cm sec^{-1} depending on season and location. Bottom deposits generally consist of rounded rocks (10 - 50 cm dia), sand and sediments. Discharge rates are generally low, 0.3 - 0.4 m^3 sec^{-1} in July and August, respectively. Water temperatures are at or near 0° C from November to May, and gradually increase to 8.0 - 10.5° C during July and August.

Mine waste water, containing heavy metals, cyanide and secondary treated sewage enter the river through several seepage streams adjacent to waste treatment ponds. Collections were made at nine stations situated 0.0 to 30 km downstream from this area and at two upstream control stations (Table 1). Water for chemical analysis was collected monthly 5 cm below the surface at each station, and preserved and analyzed as outlined in LIVELY (1974). Sediments for chemical analysis were collected in duplicate during August from the upper 3 cm of substrate at

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each site using a coring tube lined with plastic. In the laboratory the samples were digested with perchloric and hydroflouric acids and the concentrations of most metals determined by plasma emission spectrometry. In addition, mercury levels were measured by flameless atomic absorption spectrometry and arsenic by hydride generation followed by colourimetric

TABLE 1

Location of stations and annual mean and range in concentrations of copper and cyanide in the waters of the Flat River. Seasonal data are not available for stations D and E.

Station	Distance from Waste Source (km)	Copper (mg L^{-1})	Cyanide (mg L^{-1})
A (Control)	5.2	<0.005 (-)	<0.005 (-)
B (Control)	2.5	0.005 (0.005-0.007)	<0.005 (-)
C	0.0	0.009 (0.005-0.013)	0.06 (0.05-0.25)
D	0.3	-	-
E	0.5	-	-
F	1.2	0.029 (0.006-0.067)	0.08 (0.01-0.13)
G	5.0	0.005 (0.005-0.009)	0.005 (0.005-0.012)
H	9.0	<0.005 (-)	<0.005 (-)
I	9.5	0.005 (0.005-0.008)	<0.005 (-)
J	20.5	<0.005 (-)	<0.005 (-)
K	30.0	<0.005 (-)	<0.005 (-)

determination with silver diethyldithiocarbamate.

Epipelic algae were collected at each station from three sites at a water depth of 10 - 20 cm. The sediments were sampled over an area of 25 cm^2 using an aspirator and preserved in 1% Lugol's iodine solution. Densities were determined using a volumetric subsampling technique (MOORE 1979a). The zoobenthos was collected using a Surber sampler (mesh size 0.8 mm) which was in turn enclosed in a plankton net (mesh size 0.25 mm). This permitted the filtration of both large and small particles, with little clogging of the net. All samples were collected to a depth of 15 cm into the substrate and preserved in 2% formalin prior to enumeration in the laboratory. Although invertebrate sampling methods may not produce quantitatively accurate results, the use of the same sampler in both control and impacted areas permits inter-station data comparisons. In order to facilitate and clarify the analysis of seasonal variations in impacts, biological data are presented only for the months of August and December.

RESULTS

The concentration of copper in the sediments increased sharply near the mine, reaching 1500 - 2000 mg kg^{-1} at stations E - I compared with 37 - 40 mg kg^{-1} in the control areas (Fig 1). Although similar distribution patterns were recorded for several other metals, including manganese, beryllium, iron and cobalt, the concentrations of arsenic, lead, zinc, cadmium and mercury were low throughout the river, with averages of 25, 15, 190, 1 and 0.05 mg kg^{-1} respectively. Heavy metal levels in the water were generally below detectable limits at all stations throughout the year. However, cyanide was consistently high at stations C - F near the mine; in addition, copper periodically rose to 0.07 mg l^{-1} in this area, compared with normal background levels of 0.010 mg L^{-1}. Total hardness ranged from 58 to 91 mg L^{-1} regardless of station while pH varied from 7.7 to 8.1. NO_3 - N was low (0.03 - 0.08 mg L^{-1}) throughout most of the river, but increased to 0.04 - 0.97 mg L^{-1} at stations C - I due to the input of sewage. Total phosphorus levels showed a similar pattern of variation, ranging from $<$ 0.005 - 0.010 mg L^{-1} in control and downstream areas to 0.006 - 0.018 mg L^{-1} at stations C - I.

There were 107 species of algae in the August collections of which 88 were diatoms, 14 were chlorophytes and 5 were blue-greens. The most common species were *Fragilaria capucina, Diatoma tenue, Cymbella ventricosa* and *Achnanthes minutissima*, all of which occurred abundantly throughout the study area. The total density of the epipelon varied inconsistently among the stations (Fig. 1). For example large populations ($3\text{-}5 \times 10^8$ μm^3 cm^{-2}) were recorded in the control areas and at station D - I, whereas at stations C (immediately adjacent to the waste source), J and K, densities varied from 1.0 to 2.5×10^8 μm^3 cm^{-2}. A similar inconsistent trend was recorded for the dominant species (Fig. 1).

There were fewer species (96) in the December collections, a result of a reduction in the number (4) of chlorophytes. The predominant alga, *Diatoma tenue*, represented 20 to 62% of the flora, depending on station, and was followed in abundance by *Ceratoneis arcus, Fragilaria vaucheriae* and *Achnanthes minutissima*. The density of the epipelon near the mine was much higher than that recorded during the summer averaging 0.9×10^9 μm^3 cm^{-2} (range 0.5 - 1.3×10^9 μm^3 cm^{-2}) at stations D - G. The control area populations were, however, relatively small (range 0.1 - 0.45×10^9 μm^3 cm^{-2}), as also noted for stations H - K, where densities of 0.05 - 0.15×10^9 μm^3 cm^{-2} were recorded.

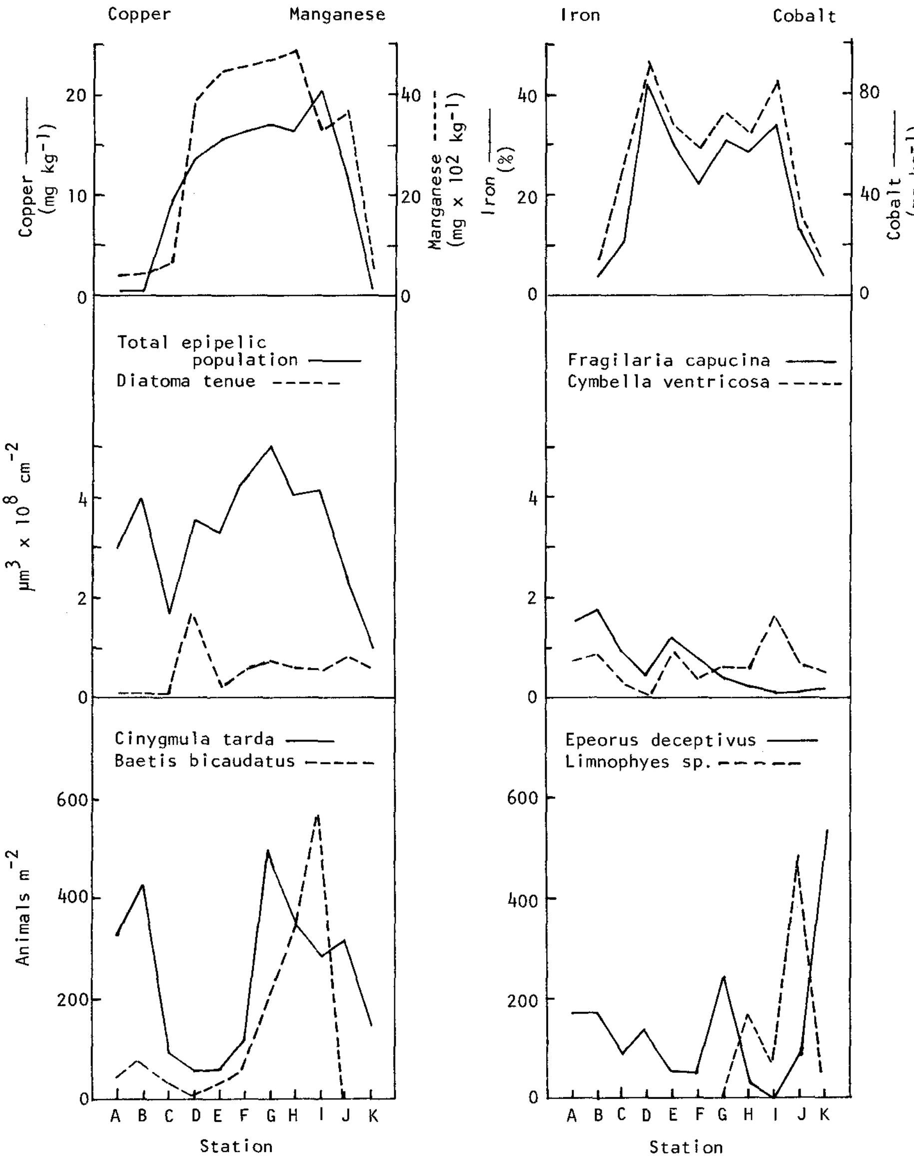

Figure 1. Concentrations of metals in the sediments and densities of benthic algae and invertebrates during August 1975 at stations A-K in the Flat River.

There were 29 species of benthic invertebrates found in the August collections, the most common of which were the mayflies Cinygmula tarda, Baetis bicaudatus and Epeorus deceptivus (Fig. 1). While these species occurred throughout the system, the predominant chironomids (Orthocladius sp., Limnophyes sp.) were restricted to stations below the mine. Other organisms, such as stoneflies and blackflies were rare in all collections. The densities of C. tarda and B. bicaudatus were generally high in areas A and B but decreased at stations C - F near the mine. Both species showed considerable recovery in areas G to I, where they reached their maximum population size. Several other common taxa such as E. deceptivus and Limnophyes sp. did not show a consistent pattern of abundance among control and impacted stations (Fig. 1).

The species composition of the benthos during December differed considerably from that observed in the summer. Cinygmula tarda was almost completely absent while Baetis bicaudatus occurred in much reduced numbers. These organisms were largely replaced by immature stoneflies, with Alloperla coloradensis being predominant throughout the study area. The density of this organism averaged 620 animals m^{-2} at stations A and B, but decreased to 210 to 590 m^{-2} in areas C to F. The population waxed at station G to an average of 1100 animals m^{-2} and remained well developed (230 - 730 m^{-2}) in downstream areas. Arcynopteryx signata, another stonefly, was second in abundance in all collections, except at station E where it was the predominant species at 500 animals m^{-2}.

DISCUSSION

One of the main points of this study is that the impact of mine wastes varied significantly with the species under investigation. On a broad scale, algae were less effected by the discharges than insects; this feature, which has been noted in a number of temperate zone systems (EPA 1973, 1976)is probably related to the fact that many forms possess the ability to physiologically adapt to high heavy metal levels (STOKES 1975, STOCKNER & ANITA 1976, MOORE et al.1979). In addition since cyanide specifically inhibits cytochrome oxidase systems involved in oxygen metabolism, it is less toxic to plants than to animals (EPA 1973). On a more specific basis, the reductions in densities of C. tarda and B. bicaudatus near the mine were probably related to high concentrations of copper and cyanide in the water, which contrasts with the negligible impact of these pollutants on E. deceptivus and Limnophyes sp. While the actual basis for the ability of the latter two organisms to survive in contaminated water is not known, such information is essential in the designation of potential indicator species. Clearly the delineation of impact in the Flat River zones depended largely on the use of organisms which did not possess the ability to adapt to polluted waters. The same may be said about epipelic algae, none of which showed a well-defined or individualistic response to contaminants in this study. It is also important to point out that in a broad sense, specific responses will almost certainly depend on the relative proportions of various contaminants, a reflection of synergistic and/or antagonistic interactions; hence species-related variability in response to the toxic action of mine wastes may fluctuate from site to site. This may in turn account for some of the difficulties which have been encountered in designating indicator species status to organisms (CAIRNS et al. 1972, MOORE 1979b).

The second main point of this study is that the impact of chemical discharges on the receiving environment showed clear seasonal variations. For example, the densities of the main invertebrate species near the mine in December were considerably higher than those recorded in August. Since the concentrations of copper and cyanide were roughly similar regardless of time of year, low temperatures may have reduced the toxicities of both pollutants during the winter (EPA 1973, 1976). On the other hand, there were considerable differences in the species composition of the assemblage (a reflection of differential emergence) which could account for the observed variations in densities among species. At present, it is not known why algal populations were particularly large near the mine during the winter. Low light and temperature levels, which reduced primary production at the control stations, should have had a similar effect near the mine. Perhaps, the release of treated sewage into the river enhanced heterotrophic growth; this explanation would account the differential development among stations at a time when autotrophic growth is normally limited. It should also be pointed out that since most metals interfere with algal growth by blocking phosphorus uptake (PLANAS & HEALEY 1978), the input of nutrients into the river may have counteracted the toxic effects of the waste discharges. This would also account for the ineffectiveness of algae in delineating the zone of impact.

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