The Use of Wetlands To Remediate Gold Mine Effluent

by

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Declaration

The research presented in this thesis was carried out by the author and has not been previously submitted for credit towards any degree or diploma. The work of others, where included, has been appropriately cited.

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Abstract

The effluent from Musselwhite Mine was examined to determine the improvement to water quality as it passed through a treatment wetland prior to final discharge to the receiving water body. Potential toxicity problems from the discharge of Al, Cu, Fe, Mn, Ni, Zn, As, Hg, and CN into the mine's industrial treatment wetland were assessed by an examination of sediment/soil and uptake of these contaminants into vegetation. In total, five study sites were established along the entrance of wetland to the receiving environment. Two of the study sites were positioned within the treatment wetland and a final site in the receiving environment each with corresponding control sites. The treatment wetland was proven to reduced the loadings of Cu, Ni, As and Al by absorption into the sediment or uptake by plants. Concentrations of Fe, Mn and Zn were released by the wetland and were slightly elevated in the discharge water.

The release of heavy metals from acid mine drainage from the tailings area of the closed Dona Lake Mine (51°42' N 90°6' W) was examined in the second part of the study. Potential toxicity due to the release of Al, Cu, Fe, Mn, Ni, Zn, As, Hg, and CN from the mine's tailings was assessed by an examination of sediment/soil sorption and uptake by vegetation. Four sites were established extending from the tailings impoundment to the receiving environment. Heavy metals released from acid mine drainage caused corresponding elevated metal concentrations in the sediment/soil present at the Dona Lake Mine. Total metal levels in the plant tissue were not elevated when all species were considered simultaneously at the various sites, but increases in metal concentrations in plant tissue did occur when only *Typha latifolia* was considered.

Reclamation of gold mine tailings waste through the development of wetland vegetation covers to determine the control of heavy metal release was examined in the third part of the study. Using a microcosm approach, the potential toxicity of the release of Al, Cu, Fe, Mn, Ni, Zn, As, Hg, CN and SO_4 from the mine's tailings was assessed through an examination of changes in water and sediment/soil concentrations of these compounds, and uptake by vegetation. Four treatments were used consisting of : 1) a control treatment that consisted only of the tailings and water with no added vegetation, 2) a tailings only treatment with added vegetation, 3) a fertilized tailings treatment which included the addition of fertilizer pellets and vegetation and 4) an organic fertilized treatment (15 cm cap of organic soil plus fertilizer pellets) and vegetation. The ability of the vegetative cover to reduce concentrations of contaminants in the water column was most successful in the organic plus fertilizer treatment. The species considered most suitable in terms of survival and limited heavy metal uptake was *Nuphar variegatum*.

General Introduction

The importance of mining cannot be overestimated. It is the world's oldest and most important industry second, only to agriculture (Down and J. Stocks, 1977). Metals and minerals produced from mining activities are essential products in modern society. Mining is the foundation upon which all other industries are dependent since the prime products of mining provide manufacturing industries with fuel and metals for power and machinery (Hawkins, 1919). All material items present in modern society are either directly a mineral product or produced with the aid of mineral derivatives (Ripley et al., 1996).

Mining is not a harmless activity to the environment. It is impossible to extract and process minerals without affecting the land, water and air to some degree. Primarily the emphasis is placed on the environmental impact arising from the presence of a mine (Turney and Thomson., 1993). The goal and commitment of today's mining companies is to minimize the disruption of the environment during exploration and production and to maximize the land rehabilitation upon closure of the mine site (Hester and Harrison., 1994).

One of the most important ecosystems are wetlands (Mitsch and Gosselink., 1993). The term "wetland" describes ecosystems that are transitional between uplands and deeply flooded lands. The species diversity present in wetlands is attained through the hydrology, soils, climate, and regional plant communities. Wetlands are defined as land where the water table is at (or above) the ground surface long enough each year to maintain saturated soil conditions and the growth of related vegetation (Reed et al., 1995). It has been found that wetland systems may act as efficient water purification systems and nutrient sinks. Effective removal of particulate matter may be achieved from long retention times and large sediment surface area in contact with the flowing water (Knight et al., 1993).

Wetland soils are the foundation for all biotic and abiotic components that exist in wetlands (Kadlec and Knight., 1996). A lack of oxygen induced by flooding characterizes many wetland soils. In flooded soils, the oxygen diffusion is nearly 10,000 times slower than in aerobic soils (Pierzynski et al., 1994). The chemical constituents in flooded wetlands may be concentrated or diluted depending on the chemistry of the water, physical and chemical nature of the soils and the surrounding environment (Armstrong, 1978). These soils support plant and microbial growth which provide the substrates necessary for water quality enhancement in wetland treatment systems. The wetland treatment system may have inadequate plant cover if soils are unsatisfactory for plant or microbial growth.

A wide assortment of plants occur naturally in wetland environments. Wetlands are home to a wide assortment of plant species due to the unique environment induced by the presence of ample water (Mitsch and Gosselink., 1993). The dominant structural element of most wetland treatment systems are the wetland macrophytes. A successful treatment wetland design relies on a basic understanding of the growth requirements and characteristics of these wetland plants. An important factor in treatment wetlands is the mineral composition of the plants. This is significant since the uptake and release of chemical compounds by plants changes water quality. Also, the mineral nutrition of plants must be adequate for their survival and growth (Kadlec and Knight., 1996). The main contaminants associated with mining activities are those produced as a byproduct of extracting the metals (Cyanide (CN), and sulphate compounds) and heavy metals that are released with newly exposed parent material.

The term "heavy metals" usually refers to those metals between atomic number 21 and 84 which occur either naturally or from anthropogenic sources in natural waters (Forstner and Wittmann., 1981). Aluminum and the metalloids are often included in this class of pollutants. Depending on their concentration and chemical speciation, these metals can be toxic to aquatic organisms (Knezek and Ellis., 1980). Since the total heavy metal is conserved, these pollutants may be considered a greater problem than organic chemicals because of the metals persistence in the environment (Schnoor, 1996).

In this study, the overall objective is directly concerned with the impact that mining has upon the external environment and the ways by which this impact may best be reduced. The specific objectives were: (1) to evaluate the effectiveness of an established wetland as a water treatment system for contaminants discharged from an active Musselwhite gold mine, (2) to examine the impact of a closed mine on the wetland environment downstream from the Dona Lake Mine site and (3) to determine the effectiveness of a vegetative cover added to gold mine tailings to reduce concentrations of heavy metals and other compounds in the water column in a controlled experiment.

Chapter 1

The Effectiveness of a Treatment Wetland on the Removal of Heavy Metals from Gold Mine Effluent at the Active Musselwhite Mine

Introduction

Recent environmental research in the mining industry has focused on the environmental challenges created by the discharge of industrial wastes which can overwhelm receiving waters (Kadlec and Knight., 1996). To deal with the concentration of pollutants and to protect receiving waters, the use of wetlands has been proposed as an inexpensive natural method of providing treatment (Gusek, 1995).

The extent of wastewater purification by natural wetlands through a complex variety of biological, physical, and chemical processes is well documented (Spangler et al., 1976, Tilton et al., 1976). The aquatic plants are the apparent biological component of the wetland ecosystem that removes the pollutant, although the vegetation alone cannot always account for the total pollutant removed (Stephenson et al., 1980, Nichols, 1983). The additional mechanisms of pollutant removal in wetland systems include bacterial transformations and physico-chemical processing, adsorption, precipitation and sedimentation (Chan et al., 1982).

Uptake of elements in aquatic plants can occur through aerial and submerged leaves, stems or roots from water, and roots from interstitial solutions in sediments (Brix, 1989). Plants roots obtain the essential elements from the interstitial water in the soil while carbon (C) and oxygen (O_2) are obtained from the atmosphere (Ripley et al., 1996). The translocation of O_2 from the shoots to the roots helps stimulate decomposition in anaerobic environments (Armstrong, 1964).

Although the metals such as Cu, Fe, Mn, and Zn are essential plant micronutrients, excess amounts of these metals may be highly toxic to plants (Krauskopf, 1972). With slight increases in concentrations, elements may go from being a micronutrient essential to plant growth to an element toxic to plants. Furthermore, a plant's capacity for selective uptake is often insufficient to prevent high concentration of metal ions from being absorbed (Down and Stocks, 1977). The absorbed metals can disrupt active metabolic sites within the cells and inhibit metabolism (Fitter and Hay., 1981).

Different contaminants are present in the effluent from mines and may or may not be taken up by plant species present in the receiving environment. The net toxicity of metals in the environment depends on several factors. First, the speciation of the metal, free aqua metal ions are toxic to aquatic biota and the complexed metal ions are not (Schnoor, 1996). The relative degree of adsorption or binding to particles in natural waters are directly affected by the chemical speciation. Second, different plant species vary in their susceptibility, while the chemical composition in which the metal exists is also important since it controls the availability of the metal to the plant (Wild, 1988). Thirdly, potential existence of synergistic interactions between contaminants in aquatic environments, for example, the toxicity of Cu and Zn together is greater than their individual toxicities (Down and Stocks, 1977).

This study examines the potential development of environmental toxicity problems from the mine's industrial discharge of Al, Cu, Fe, Mn, Ni, Zn, As, Hg, and CN into a wetland treatment system. The objectives of the study were to assess the impact of effluent from the gold mine, Musselwhite on the treatment wetland, and to determine if the wetland was able to reduce the concentrations of these contaminants on the final receiving water body.

Methods

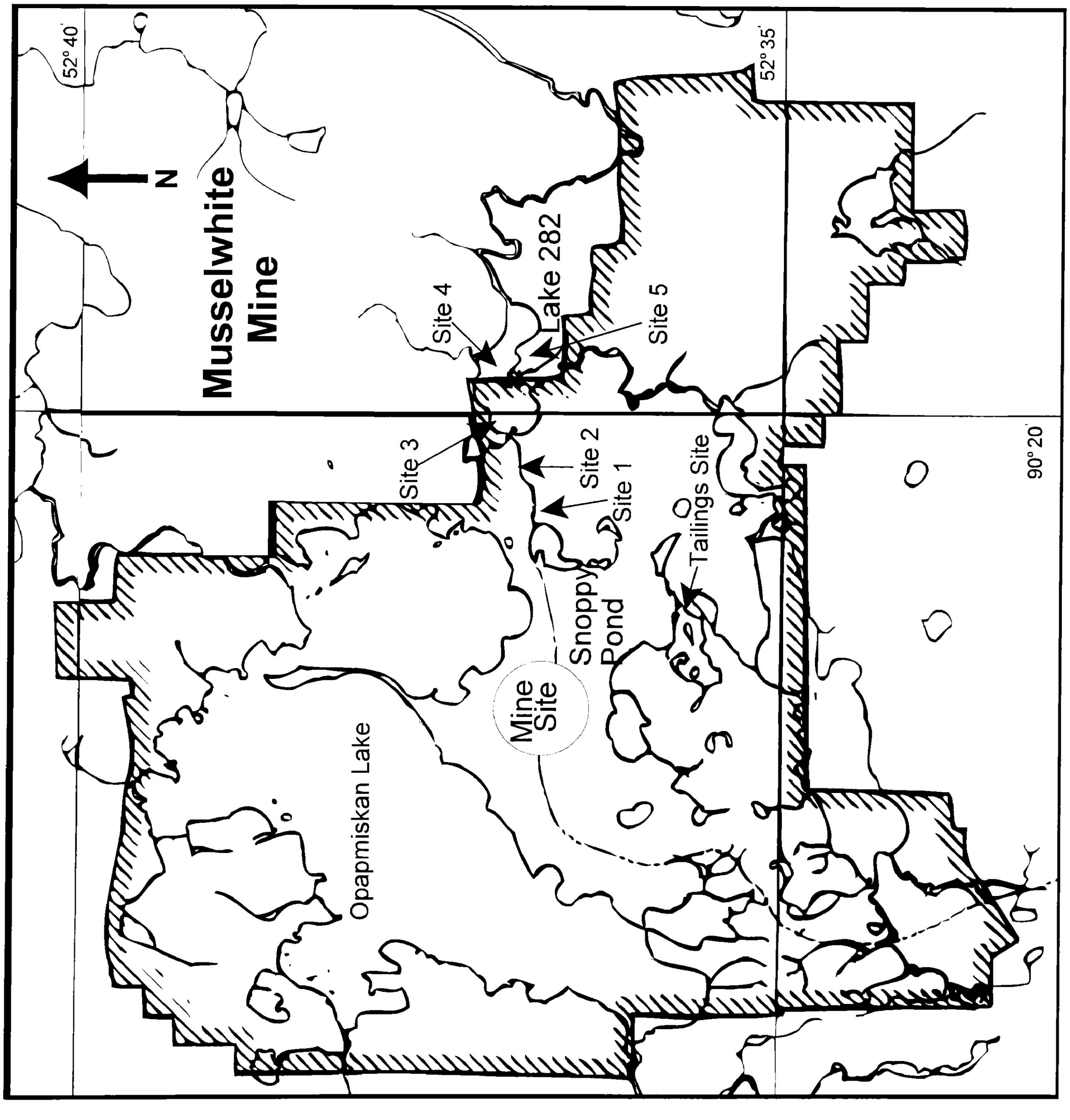
Field Site

The Musselwhite Mine, located approximately 130 km north of Pickle Lake, Ontario, Canada (52°37' N 90°24' W) and owned by Placer Dome North America Ltd. was the focus for this study (Figure 1). The Musselwhite gold mine construction began in November, 1995 and it officially opened on July 23, 1997. The open pit and underground establishment was the first mining project in Canada to require review under the Canadian Environmental Assessment Act (CEAA) and the Environmental Effects Monitoring (EEM) program. Several environmental control plans were implemented which included: cyanide recycling within the mill, treatment of tailings slurry using the Inco SO_2/Air process, recycling of tailings pond supernatant to the mill to be used as process water, and directing of pit water, wastewater run off, and site run off, to the polishing ponds for treatment prior to release (Anonymous, 1995). A treatment wetland was established by the expansion of the Snoppy creek, to allow for final treatment with retention times between two to three weeks before release to the receiving environment, Lake 282.

Five study sites were selected ranging from the polishing pond, Snoppy Pond, to the receiving environment, Lake 282. The five study sites were chosen in the following areas; Area 1 - a moderately rich fen, located in the area immediately adjoining and draining the polishing pond (Snoppy Pond), Area 2 - an intermediate swamp, located along Snoppy Creek, Area 3 - open water marsh, located downstream from the outlet of the creek draining Snoppy Pond to Lake 282, Area 4 - the control station for Area 3, located upstream from the mine discharge in the same ecotype at the south end of Lake 282, and Area 5 - the control station for Area 1 and 2, located upstream from the mine discharge in the same ecotype at the

Figure 1

Location of Musselwhite Mine and the Musselwhite Mine study site locations (a) Site 1, (b) Site 2, (c) Site 3, (d) Site 4, (e) Site 5.


Musselwhite Mine
N
52° 40'
52° 35'
90° 20'
Lake 282
Site 4
Site 5
Site 3
Site 2
Site 1
Snoppy Pond
Mine Site
Tailings Site
Opapmiskan Lake

south end of Lake 282. The sampling took place in mid June, July, September 1997 and in early June, July, and August 1998.

Sampling and Laboratory Procedures

The data collection procedures for the vegetation followed a modified version of Harris (1993). At each of the areas a permanent sampling area, 10m x 10m in size was established. Each sampling period the vegetation and sediment samples were collected from a randomly selected $1m^2$ quadrat. At each area, six dominant plant species were sampled (Table 1) and two sediment/soil samples (upper 20 cm layer) were collected in the middle of the quadrat. Quality Assurance Quality Control (QA/QC) measures were implemented for the sampling procedures of both the sediment/soil and vegetation samples. For each sampling period the sediment/soil and vegetation QA/QC procedures involved obtaining a reference sample, a split sample, and a duplicate sample. Field measurements of pH, conductivity of the standing water and depth to the water table or depth of the standing water and profiles of the site quality were measured and recorded. The Municipal Industrial Strategy for Abatement (Ontario) (MISA) water data which was collected at the entrance and the discharge of the wetland, were obtained from the Musselwhite Mine Environmental Department.

Vegetation and Sediment/Soil Analysis

The vegetation and sediment/soil samples were packaged and stored on ice in portable coolers for transport to the laboratory and were kept frozen until prepared for analysis. A portion of each of the sediment/soil sample was oven dried at 80 ^{0}C in a preweighed 20 cm^3 crucible to obtain bulk density measurements. All of the vegetation and

Table 1
Predominant Vegetation Species Sampled at Musselwhite Mine

	Collection of Species					
	June		July		September/August	
Site	1997	1998	1997	1998	1997	1998
Site 1	*-Rubus pubescens* *-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Betula papyrifera* *-Picea mariana* *-Alnus incana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Betula papyrifera* *-Alnus incana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Ledum groenlandicum* *-Equisetum sylvaticum* *-Betula papyrifera* *-Alnus incana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Ledum groenlandicum* *-Equisetum sylvaticum* *-Betula papyrifera* *-Alnus incana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Ledum groenlandicum* *-Equisetum sylvaticum* *-Betula papyrifera* *-Alnus incana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Ledum groenlandicum* *-Equisetum sylvaticum* *-Betula papyrifera* *-Alnus incana*
Site 2	*-Equisetum sylvaticum* *-Maianthemum trifolium* *-Eriophorum vaginatum* *-Ledum groenlandicum* *-Picea mariana* *-Betula papyrifera*	*-Equisetum sylvaticum* *-Maianthemum trifolium* *-Eriophorum vaginatum* *-Ledum groenlandicum* *-Betula papyifera* *-Salix bebbiana*	*-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Eriophorum vaginatum* *-Betula papyrifera* *-Salix bebbiana* *-Picea mariana*	*-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Eriophorum vaginatum* *-Betula papyrifera* *-Salix bebbiana* *-Picea mariana*	*-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Eriophorum vaginatum* *-Betula papyrifera* *-Salix bebbiana*	*-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Eriophorum vaginatum* *-Betula papyrifera* *-Salix bebbiana*
Site 3	*-Calla palustris* *-Glyceria borealis* *-Equisetum fluviatile* *-Scirpus acutus*	*-Calla palustris* *-Glyceria borealis* *-Equisetum fluviatile* *-Scirpus acutus* *-Potentilla palustris* *-Iris versicolor*	*-Calla palustris* *-Equisetum fluviatile* *-Scirpus acutus* *-Glyceria borealis* *-Elatine triandra* *-Nuphar variegatum*	*-Calla palustris* *-Equisetum fluviatile* *-Scirpus acutus* *-Glyceria borealis* *-Potentilla palustris* *-Iris versicolor* *-Nuphar variegatum*	*-Calla palustris* *-Equisetum fluviatile* *-Scirpus acutus* *-Glyceria borealis* *-Potentilla palustris* *-Nuphar variegatum*	*-Calla palustris* *-Equisetum fluviatile* *-Scirpus acutus* *-Glyceria borealis* *-Potentilla palustris* *-Iris versicolor* *-Nuphar variegatum*
Site 4	*-Eleocharis smallii* *-Carex lacustris* *-Equisetum fluviatile* *-Myrica gale*	*-Carex lacustris* *-Equisetum fluviatile* *-Myrica gale* *-Carex aquatilis*	*-Carex lacustris* *-Eleocharis smallii* *-Equisetum fluviatile* *-Myrica gale* *-Nuphar variegatum*	*-Carex lacustris* *-Eleocharis smallii* *-Equisetum fluviatile* *-Myrica gale* *-Nuphar variegatum* *-Carex aquatilis*	*-Carex lacustris* *-Eleocharis smallii* *-Equisetum fluviatile* *-Myrica gale* *-Nuphar variegatum* *-Sparganium fluctuans*	*-Carex lacustris* *-Eleocharis smallii* *-Equisetum fluviatile* *-Myrica gale* *-Nuphar variegatum* *-Sparganium fluctuans* *-Carex aquatilis*
Site 5	*-Maianthemum trifolium* *-Rubus pubescens* *-Ledum groenlandicum* *-Equisetum sylvaticum* *-Larix laricina* *-Alnus incana*	*-Maianthemum trifolium* *-Rubus pubescens* *-Ledum groenlandicum* *-Equisetum sylvaticum* *-Salix bebbiana* *-Alnus incana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Alnus icana* *-Salix bebbiana* *-Larix laricina*	*-Rubus pubescens* *-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Alnus icana* *-Salix bebbiana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Alnus incana* *-Salix bebbiana*	*-Rubus pubescens* *-Maianthemum trifolium* *-Equisetum sylvaticum* *-Ledum groenlandicum* *-Alnus incana* *-Salix bebbia*

sediment/soil samples were dried in a drying oven at 50 ^{0}C for approximately 72 hours. The samples were then gently ground and passed through a 2 mm sieve.

The total metals (Al, Cu, Fe, Mn, Ni, Zn) in the vegetation and sediment/soil samples were digested in a H_2SO_4/HNO_3 matrix, diluted with deionized distilled water (DDW) and analyzed by Inductively Coupled Plasma (ICP) spectroscopy. Extractable metals (Al, Cu, Fe, Mn, Ni, Zn) in the sediment/soil samples were determined according to the methods of Lee (1986) (extraction in 0.1M HCl) and analyzed by ICP spectroscopy.

The total cyanide in the vegetation (1.0 g sample) and sediment/soil (2.5 g sample) were extracted for 16 hours in 50 mL 1% NaOH, filtered by gravity with Whatman 1 filter paper, diluted with DDW and filtered through a solid phase extractor and analyzed by a Skalar Autoanalyzer SAN-System. A nitrate correction was implemented to determine the correct total cyanide content in the sediment/soil samples for 1 ppm NO_3 = 20 ppb CN_T. The nitrate in the 0.50 g sediment/soil samples were extracted for 30 minutes with 50 mL of DDW, filtered by gravity with Watman 1 filter paper decanted into IC vials and analyzed by Ion Chromatography (IC).

The arsenic analysis for the vegetation and sediment/soil samples were digested in H_2SO_4/HNO_3 matrix to convert all forms of As to As (IV). The As in this form was measured by Hydrides Generation Atomic Absorption Spectroscopy at Accurassay Laboratory in Thunder Bay, Ontario Canada.

The mercury analysis for the vegetation and sediment/soil samples were digested in $HCl/HNO_3/KMnO_4$ matrix to extract Hg. After digestion the excess $KMnO_4$ was reduced with hydroxilamine hydrochloride and the concentration of Hg in digest was determined by Cold Vapor Hydrides Generation Atomic Absorption at Accurassay Laboratory.

All of the procedures followed standard operating procedures of LUEL, which were developed from the U.S. Environmental Protection Agency (EPA), Standard Methods of APHA, Environment Canada and the Ontario Ministry of the Environment.

Data Analysis

The computer program SPSS 7.5 standard version statistical package (Windows, 1996) was used for the data analysis. All of the data were examined for outliers along with presence of skewness and kurtosis. The means of the monthly MISA water data at the entrance and discharge points of the wetland in each year were examined statistically. In order to determine the effectiveness of the treatment wetland a one way analysis of variance (ANOVA) was preformed on the means of the monthly MISA water data to compare levels at the entrance and discharge of the wetland in each year. The metal mean of the two sediment/soil samples and the metal mean of the six vegetation species collected along with a metal mean of three common vegetation species (*Maianthemum trifolium, Equisetum sylvaticum and Ledum groenlandicum*) at sites 1, 2 and 5 were utilized for the statistical analysis. To determine the effectiveness of the wetland, a two way ANOVA was preformed on the elements in both the sediment/soil and vegetation to assess if there were any significant differences with respect to site and time. A Least Significance Difference (LSD) paired comparison test was preformed on the vegetation and sediment/soil if the ANOVAs were significant among sites to determine which site differs from each other.

Results

Water Analysis

Monthly mean total metal concentrations in the water at the entrance and exit of the wetland provided by the MISA data during 1997 and 1998 are shown in Figure 2. In 1997, only total Cu and Ni (Figure 2) were significantly decreased by the wetland. By 1998, significant differences existed in a decrease in total As, Al and Cu and significant increase in total Fe and Zn upon the exit of the treatment wetland (Figure 2).

The properties of the water measured in the field at the five sites during 1997 and 1998 are shown in Table 2. The pH of the water ranged from 6.63 to 7.46 among the sites (Table 2). Depth of water and conductivity levels of sites 1 and 2 increased from 1997 to 1998 while the controls sites 4 and 5 were lower in comparison (Table 2). The sites not contained within the treatment wetland demonstrated a decrease in depth of water from 1997 to 1998 (Table 2).

Table 2
Properties of Water Reported of Each Site at Musselwhite Mine

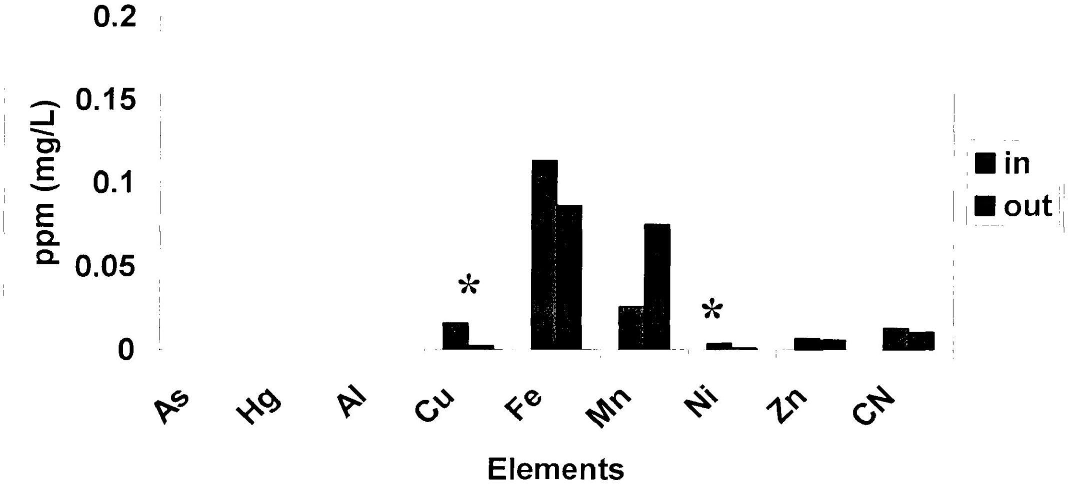
		pH			Conductivity $(1/\Omega)$			Depth (cm)		
Site	Year	June	July	Sept/Aug	June	July	Sept/Aug	June	July	Sept/Aug
1	1997	6.63	6.96	6.71	183.8	399.8	433.8	2.6	9.6	4.8
	1998	6.89	6.65	7.14	476.6	458.0	508.5	17.0	17.2	9.4
2	1997	6.40	6.63	6.90	172.5	281.4	466.8	2.2	16.4	17.6
	1998	6.64	6.72	6.86	620.8	385.6	1036.8	20.0	18.0	12.6
3	1997	6.78	6.70	6.97	33.5	89.1	95.9	39.2	29.8	7.2
	1998	6.76	6.81	6.99	123.9	79.7	298.5	23.4	16.2	11.6
4	1997	6.30	6.91	7.03	36.4	52.7	45.4	35.6	37.6	6.2
	1998	6.57	6.52	6.90	68.8	58.1	38.4	28.5	15.4	7.2
5	1997	6.41	6.93	7.10	118.9	50.3	46.4	7.0	4.6	6.0
	1998	6.51	7.44	7.46	109.1	77.6	70.9	5.0	1	1

[1] depth of standing water is a mean of four corners and centre of quadrate.

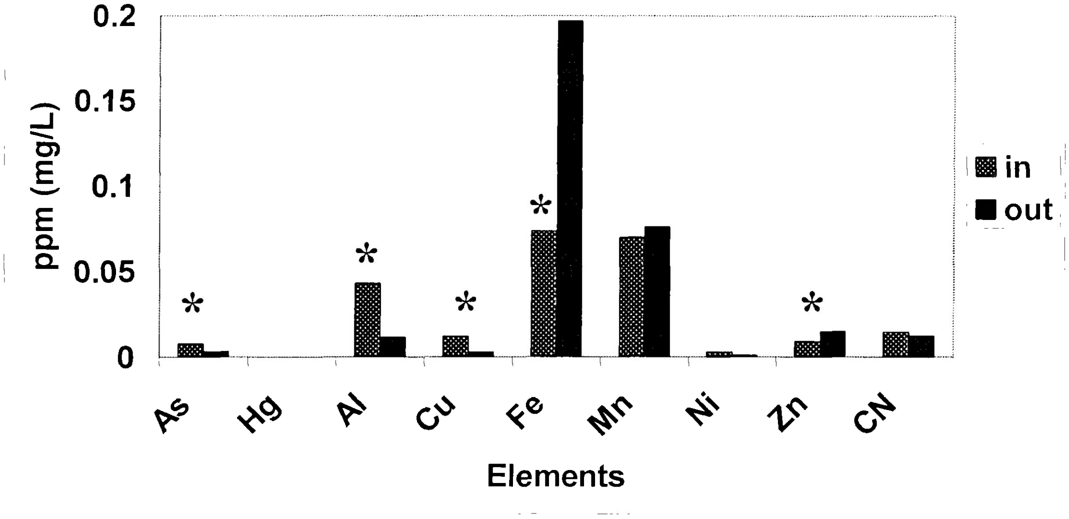
Figure 2

Total metal concentrations in the water at the entrance and discharge from the treatment wetland in (a) 1997 and (b) 1998. The * indicated statistically different at $P < 0.05$. The elements which were below detection limits are not presented.

a) Total Metal Concentration in the Water at the Entrance and Discharge of the Treatment Wetland in 1997



b) Total Metals Concentration in the Water at the Entrance and Discharge from the Treatment Wetland in 1998



Sediment/Soil Analysis

Bulk density and pH of the sediment/soil at the five sites during 1997 and 1998 are shown in Table 3. Paste pH of the sediment/soil ranged from 5.8 to 7.9 at the five sampling sites (Table 3). Bulk densities of the sediment/soil were similar over the season and between years in each site (Table 3). Site 3 had substantially lower bulk density in comparison to the remaining sites (Table 3).

Table 3
Properties of Sediment/Soil of Each Site at Musselwhite Mine

		Paste pH			Bulk Density (g/cm^3)		
Site	Year	June	July	Sept/Aug	June	July	Sept/Aug
1	1997	6.9	7.0	7.2	0.129	0.178	0.117
	1998	6.9	6.9	7.0	0.135	0.134	0.147
2	1997	7.2	7.3	7.9	0.158	0.145	0.255
	1998	7.0	7.0	6.6	0.164	0.135	0.235
3	1997	6.7	6.4	6.9	0.093	0.081	0.122
	1998	6.2	6.5	6.0	0.104	0.118	0.101
4	1997	5.9	6.5	5.8	0.122	0.09	0.099
	1998	5.7	6.0	5.6	0.146	0.09	0.123
5	1997	7.2	7.4	7.2	0.175	0.213	0.160
	1998	7.0	6.8	6.7	0.148	0.183	0.180

[1] bulk density and paste pH are means of the samples collected at each site

Concentrations of total metals (As, Cu, Fe, Ni, Zn) in the sediment/soil present at the terrestrial sites for both 1997 and 1998 were generally higher at site 1 and/or 2 with lower concentrations at the control site 5 (Figure 3). In both 1997 and 1998, concentration of total metals (As, Fe, Mn) in the sediment/soil of the open water sites was generally higher at site 3 with lower concentrations at the control site 4 (Figure 3).

Mean total metal concentrations in the sediment/soil at the five sampling sites during 1997 and 1998 are shown in Table 4. One way ANOVA revealed statistical differences among all sites for total metals with the exception of Hg and Mn in 1997 and Mn and Zn in 1998 (Table 4). Arsenic was statistically different in both years among site 1 with site 2 and

5, in 1998 Cu, Fe, Mn and Ni were statistically different (Table 4). Total Hg and Ni were statistically different between sites 1 and 2 with the control site 5 (Table 4). A statistical decrease in total Al and a statistical increase in total Cu, Mn and Ni between years were shown (Table 4). Nickel statistically increased in sites 1 and 4 decreased in sites 2, 3 and 5 (Table 4). Overall the highest total metal concentrations were Al and Fe (Table 4).

Extractable metal concentrations in the sediment/soil at the five sites during 1997 and 1998 are shown in Figure 4. The concentration of extractable Cu, Mn, Ni, and Zn in 1997 and 1998, present in the sediment/soil at the terrestrial sites were generally higher at site 1 and/or 2 than the control site 5 (Figure 4). The concentration of the extractable Mn in the sediment/soil present in the open water sites were higher at site 3 than the control site 4 (Figure 4).

Mean extractable metal concentrations in the sediment/soil at the five sampling sites during 1997 and 1998 are shown in Table 4. A one way ANOVA revealed statistical differences among all sites for extractable metals with the exception of Mn and CN in 1998 (Table 4). The only extractable metal which was statistically at different site 1 from site 2 and 5 was Cu in 1998 (Table 4). Terrestrial sites 1, 2 and 5 were statistically different from the open water sites 3 and 4 in extractable Al and Fe (Table 4). Statistical differences between years were demonstrated by a decrease in CN and extractable Fe and an increase in extractable Cu, and Ni (Table 4).

Vegetation Analysis

Metal concentrations in the vegetation at the five sites during 1997 and 1998 are shown in Figure 5. Manganese in 1997, Al and Ni in 1998, and As, Hg, Cu, Fe, Zn and CN in both 1997 and 1998 in the terrestrial sites vegetation were generally higher at site 1 and/or

2 than at the control site 5 (Table 5 and Figure 5). The concentrations of Fe, Mn, Zn, CN in 1997 and As, Al, Fe, Zn in 1998 in the vegetation of the open water sites were generally higher at site 3 than the control site 4 (Figure 5).

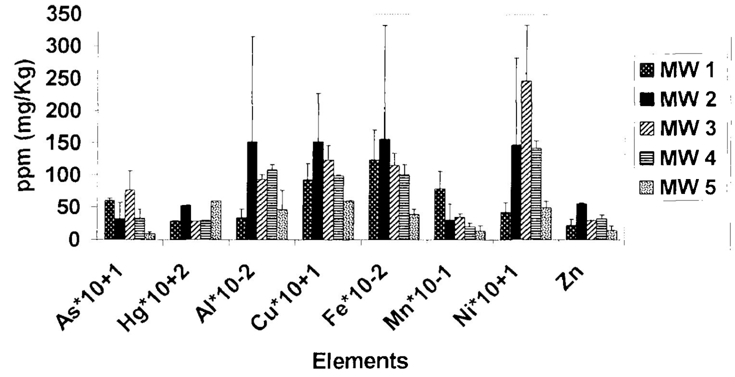
Mean metal concentrations in the vegetation at the five sampling sites during 1997 and 1998 are shown in Table 5. A one way ANOVA revealed statistical differences among sites for all metals with the exception of As and Ni in 1997 and Cu, Ni and Zn in 1998 (Table 5). Statistical differences between years were demonstrated by a decrease in total Hg and an increase in CN and total Cu, and Zn (Table 5). The total metals Al, Fe, and Zn in 1998 are statistically different in sites 1 and the control site 5 (Table 5). For all of the metals there was no statistical difference between the open water sites 3 and 4 with the exception of Fe in 1998 (Table 5).

Mean metal concentrations in the *Maianthemum trifolium*, *Ledum groenlandicum* and *Equisetum sylvaticum* at the terrestrial sites 1, 2 and 5 during 1997 and 1998 are shown in Table 6. Statistical differences between years were demonstrated by an increase in total Cu, and Zn (Table 6). Arsenic and Fe in 1997 and Al, Cu and Fe in 1998 (Table 6) were statistically different among the five sites. The metals Al, Cu and Fe present in the common vegetation species of site 1 were statistically different from site 5 (Table 6). Metal concentrations at site 1 were higher with lower metal concentrations at the control site 5 (Figure 5).

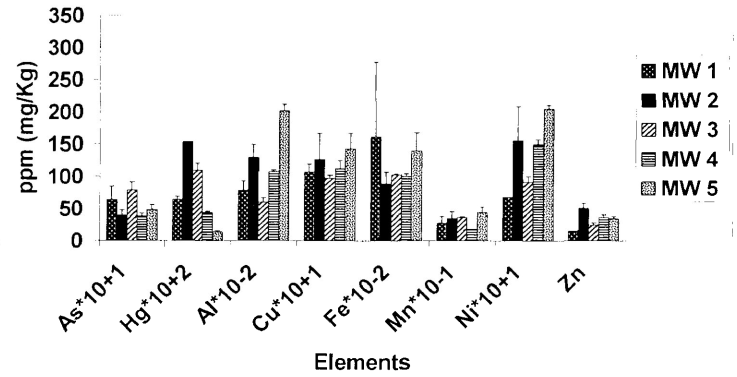
Figure 3

Treatment means of the elements of the Musselwhite sediment: (a) June 1997 total metals in sediment/soil at Musselwhite sites, (b) July 1997 total metals in sediment/soil at Musselwhite sites, (c) September 1997 total metals in sediment/soil at Musselwhite sites, (d) June 1998 total metals in sediment/soil at Musselwhite sites, (e) July 1998 total metals in sediment/soil at Musselwhite sites, (f) August 1998 total metals in sediment/soil at Musselwhite sites.

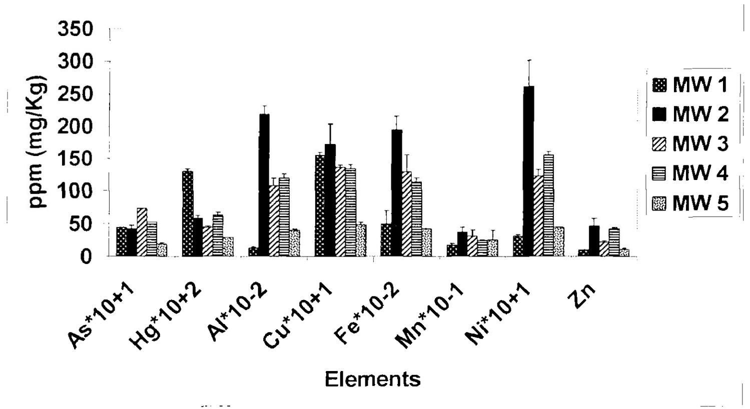
a) June 1997 Total Metals in Sediment/Soil at Musselwhite Site



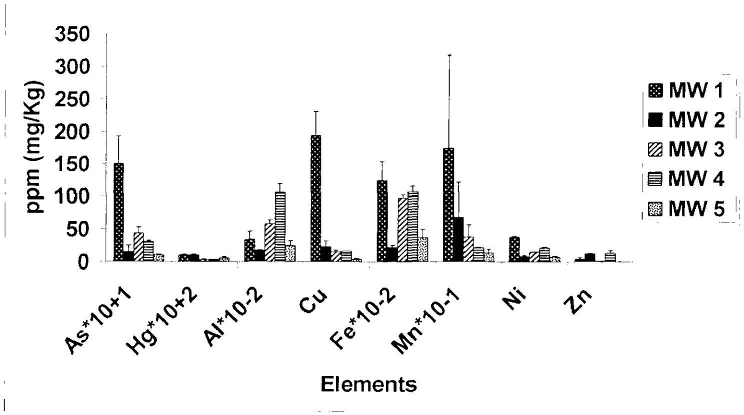
b) July 1997 Total Metals in Sediment/Soil at Musselwhite Site



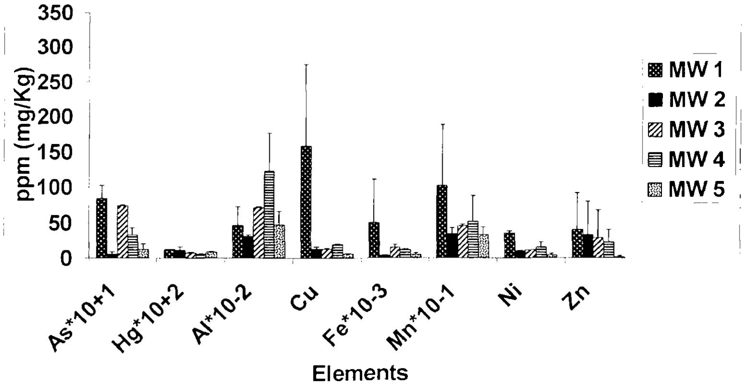
c) September 1997 Total Metals in Sediment/Soil at Musselwhite Site



d) June 1998 Total Metals in Sediment/Soil at Musselwhite Site



e) July 1998 Total Metals in Sediment/Soil at Musselwhite Site



f) August 1998 Total Metals in Sediment/Soil at Musselwhite Site

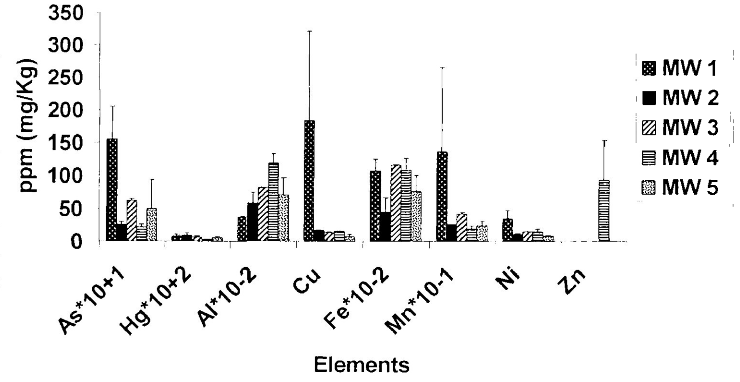


Table 4

Means of Element Concentrations (ppm) in Sediment/Soil at Musselwhite Mine

Element	Year	Site Location				
		1	2	3	4	5
As	1997	5.61 a	3.76 b	7.63 c	4.11 ab	2.54 b
	1998	12.97 a	1.54 b	5.97 c	2.83 b	2.42 b
Hg	1997	0.074	0.089	0.061	0.046	0.041
	1998	0.103 a	0.105 a	0.057 b	0.039 b	0.066 b
Total Al	1997*	4154.98 a	16654.75 b	8708.44 ac	11181.94 bc	9605.20 ac
	1998	3937.52 a	3528.98 a	7045.24 b	11618.43 c	4737.50 ab
Total Cu	1997*	11.86 ab	15.02 a	11.92 ab	11.55 ab	8.42 b
	1998	178.91 a	17.59 c	13.99 c	17.52 c	5.72 c
Total Fe	1997	11170.24 ab	14637.31 a	11583.32 ab	10499.09 ab	7364.62 b
	1998	24716.76 a	3497.98 b	12235.13 ab	11223.84 ab	5523.97 b
Total Mn	1997*	414.66	347.03	344.17	211.22	277.81
	1998	1377.24 a	428.44 b	421.30 b	312.69 b	238.74 b
Total Ni	1997*	4.71 a	18.82 b	15.34 bc	14.92 bc	9.98 ac
	1998	36.21 a	9.94 bd	13.87 bc	17.34 c	6.64 d
Total Zn	1997	15.74 a	51.08 b	25.58 c	37.45 d	20.01 ac
	1998	22.25	30.24	28.77	42.74	20.78
Extractable Al	1997	56.81 ad	52.93 a	468.97 b	1476.88 c	244.92 d
	1998	71.37 a	50.75 a	545.49 b	1556.55 c	40.08 a
Extractable Cu	1997*	0.062 a	0.207 ab	0.507 b	2.77 c	0.375 ab
	1998	2.17 a	0.744 b	0.665 b	4.87 c	0.197 b
Extractable Fe	1997*	37.66 a	27.79 a	339.25 b	462.32 c	60.51 a
	1998	38.96 a	16.12 a	243.93 b	405.45 c	20.97 a
Extractable Mn	1997	211.32 ac	83.22 b	241.05 a	125.26 bc	105.73 bc
	1998	344.38	304.30	211.68	172.65	141.96
Extractable Ni	1997*	0.300 a	0.374 a	1.25 b	2.38 c	0.593 a
	1998	6.93 a	4.33 ab	4.47 ab	4.53 ab	3.85 b
Extractable Zn	1997	3.17 a	10.91 b	6.63 ab	9.86 b	2.15 a
	1998	7.60 a	7.52 a	8.73 ab	11.68 b	1.30 c
CN	1997*	0.838	0.859	0.938	1.69	1.09
	1998	-	-	0.189	0.039	0039

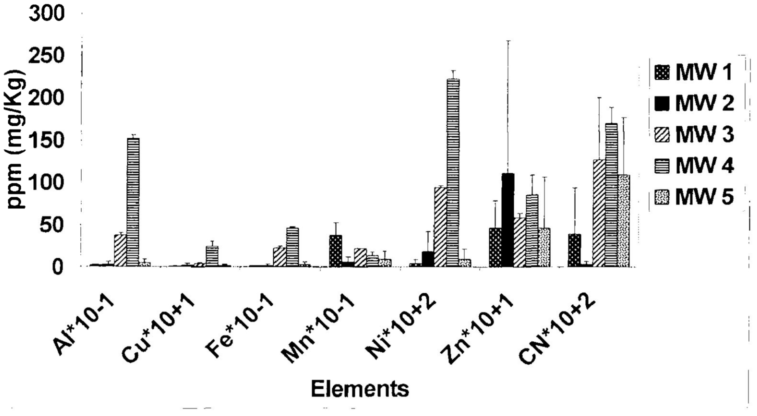
[1] Values followed by different letters in the same year were significantly different ($P \leq 0.05$).

[2] Values followed by * were significantly different between years ($P \leq 0.05$).

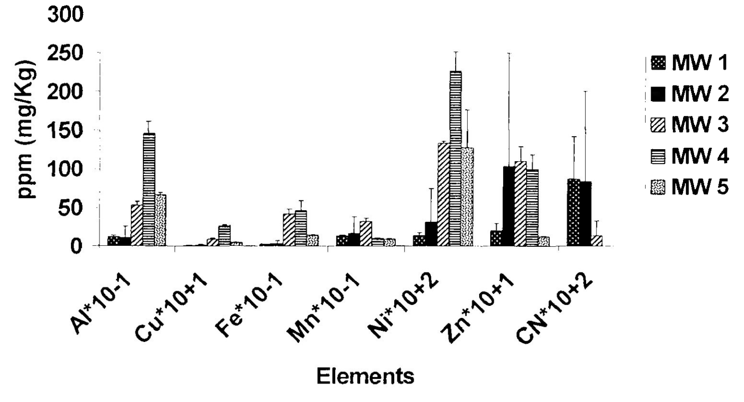
Figure 4

Treatment means of the elements of the Musselwhite sediment: (a) June 1997 extractable metals in sediment/soil at Musselwhite site, (b) July 1997 extractable metals in sediment/soil at Musselwhite site, (c) September 1997 extractable metals in sediment/soil at Musselwhite site, (d) June 1998 extractable metals in sediment/soil at Musselwhite site, (e) July 1998 extractable metals in sediment/soil at Musselwhite site, (f) August 1998 extractable metals in sediment/soil at Musselwhite site.

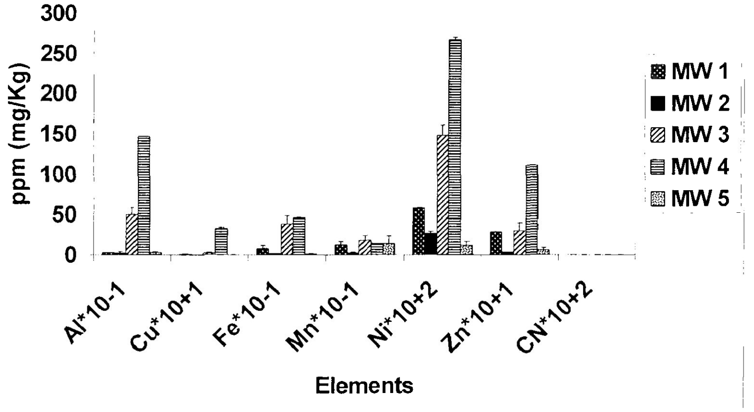
a) June 1997 Extractable Metals in Sediment/Soil at Musselwhite Site



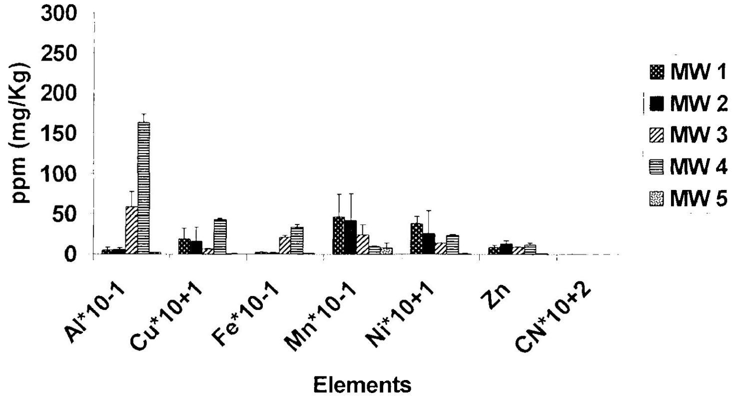
b) July 1997 Extractable Metals in Sediment/Soil at Musselwhite Site



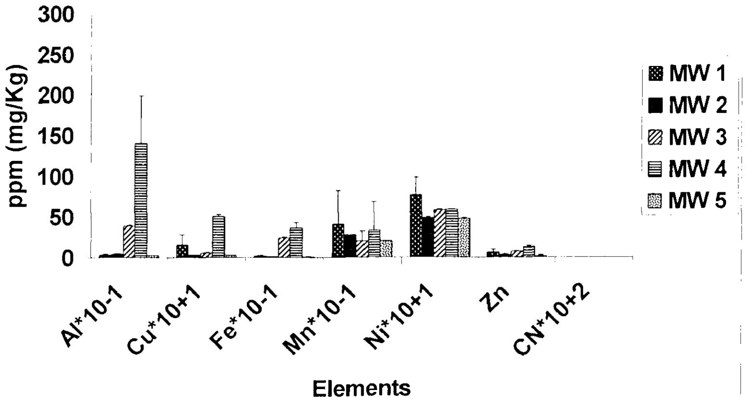
c) September 1997 Extractable Metals in Sediment/Soil at Musselwhite Site



d) June 1998 Extractable Metals in Sediment/Soil at Musselwhite Site



e) July 1998 Extractable Metals in Sediment/Soil at Musselwhite Site



f) August 1998 Extractable Metals in Sediment/Soil at Musselwhite Site

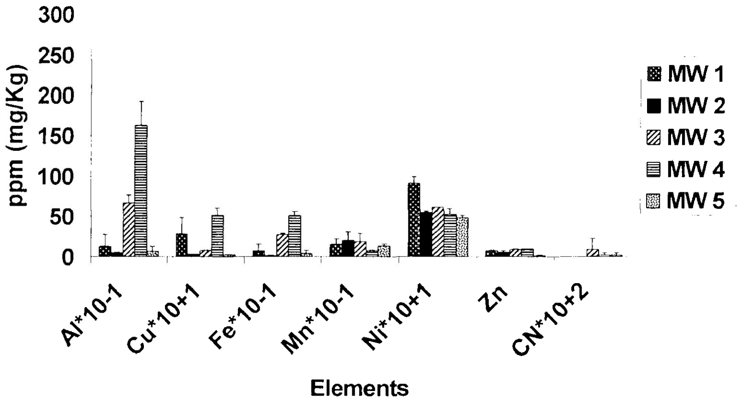
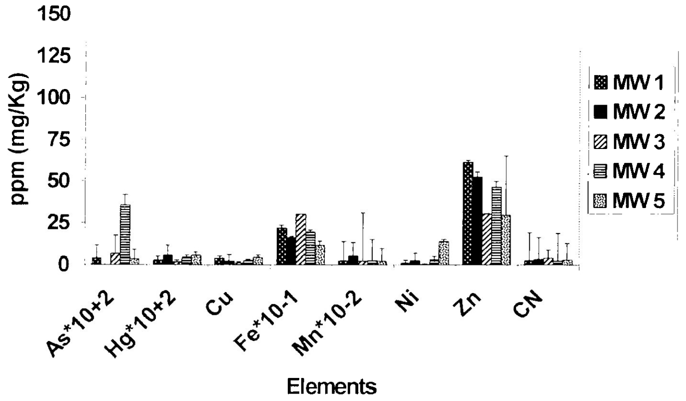


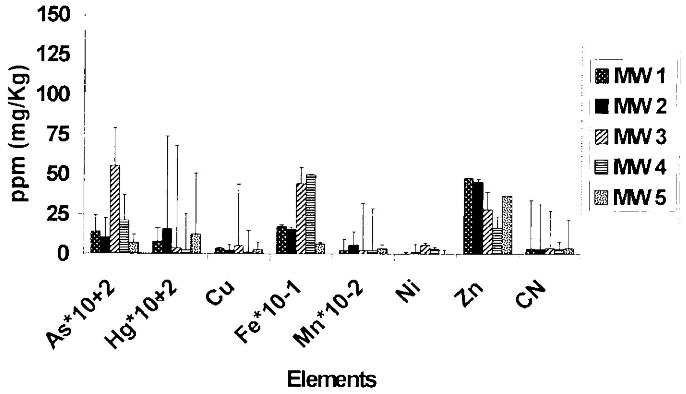
Figure 5

Treatment means of the elements of the Musselwhite vegetation species: a) June 1997 total metals in vegetation at Musselwhite sites, (b) July 1997 total metals in vegetation at Musselwhite sites, (c) September 1997 total metals in vegetation at Musselwhite sites, (d) June 1998 total metals in vegetation at Musselwhite sites, (e) July 1998 total metals in vegetation at Musselwhite sites, (f) August 1998 total metals in vegetation at Musselwhite sites.

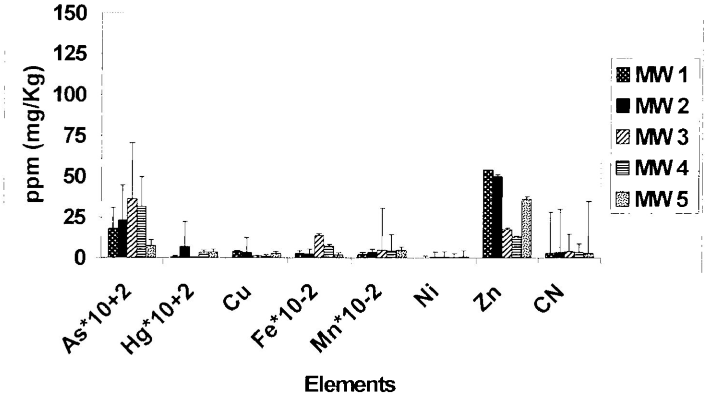
a) June 1997 Total Metals in Vegetation at Musselwhite Site



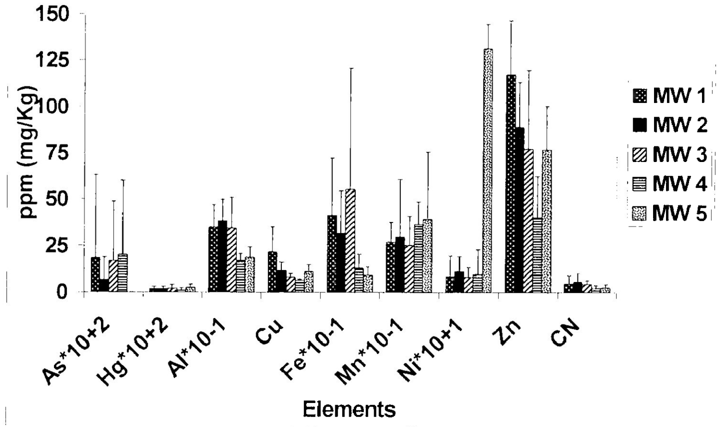
b) July 1997 Total Metals in Vegetation at Musselwhite Site



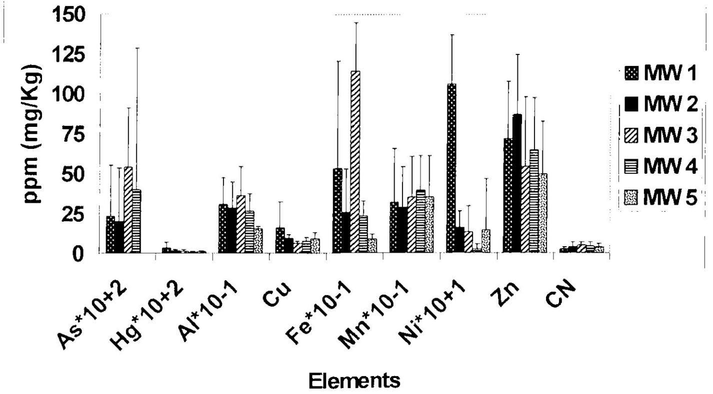
c) September 1997 Total Metals in Vegetation at Musselwhite Site



d) June 1998 Total Metals in Vegetation at Musselwhite Sites



e) July 1998 Total Metals in Vegetation at Musselwhite Sites



f) August 1998 Total Metals in Vegetation at Musselwhite Sites

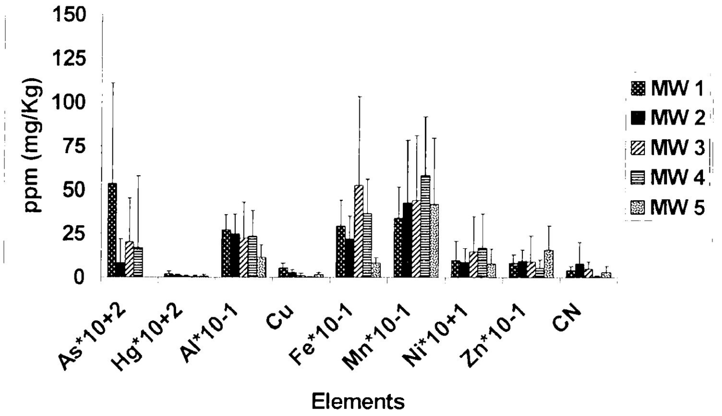


Table 5
Means of Element Concentrations (ppm) in the Vegetation at Musselwhite Mine

		Site Location				
Element	Year	1	2	3	4	5
As	1997	0.120 ab	0.166 ab	0.326 a	0.294 ab	0.059 b
	1998	0.317	0.117	0.303	0.256	-
Hg	1997*	0.038 ab	0.093 a	0.018 b	0.033 ab	0.070 ab
	1998	0.023 a	0.014 ab	0.012 ab	0.007 b	0.015 ab
Total Al	1997	-	-	-	-	-
	1998	309.25 a	305.99 a	310.27 a	228.81 ab	151.43 b
Total Cu	1997*	3.75 a	2.55 ab	2.36 ab	1.45 b	3.20 ab
	1998	14.35	7.96	5.12	4.72	7.31
Total Fe	1997	224.87 ab	185.62 ab	703.48 a	468.34 ab	121.50 b
	1998	413.83 a	265.51 ac	741.72 b	243.21 ac	87.09 c
Total Mn	1997	221.52 a	466.74 b	301.99 ab	295.14 ab	331.13 ab
	1998	309.80	338.00	349.19	447.42	388.93
Total Ni	1997	0.585	1.37	2.02	2.19	1.77
	1998	4.14	1.21	1.20	0.949	5.13
Total Zn	1997*	54.44 a	49.21 ac	25.34 b	25.35 b	34.22 bc
	1998	90.70	90.23	73.77	52.99	94.15
CN	1997*	2.83 a	3.23 ab	3.81 b	2.86 a	3.05 a
	1998	3.73 ab	5.81 a	4.78 ab	2.33 b	3.08 b

[1] Values followed by different letters in the same year were significantly different ($P \leq 0.05$).

[2] Values followed by * were significantly different between years ($P \leq 0.05$).

Table 6
Means of Element Concentrations (ppm) in Common Vegetation Species at Musselwhite Mine

		Site Location		
Element	Year	1	2	5
As	1997	0.125 ab	0.185 a	0.075 b
	1998	0.786	0.233	-
Hg	1997	0.044	0.032	0.307
	1998	0.019	0.017	0.021
Total Al	1997	-	-	-
	1998	368.64 a	307.31 a	153.87 b
Total Cu	1997*	3.19	3.69	3.46
	1998	17.89 a	8.81 ab	7.61 b
Total Fe	1997	256.39 a	204.73 ab	117.30 b
	1998	583.60 a	277.60 ab	98.47 b
Total Mn	1997	279.71	579.52	440.22
	1998	387.31	469.35	532.50
Total Ni	1997	0.659	2.55	0.995
	1998	10.55	1.22	13.15
Total Zn	1997*	27.69	30.43	23.84
	1998	70.98	71.83	62.30
CN	1997	3.87	4.74	4.43
	1998	5.09	8.79	3.80

[1] Values followed by different letters in the same year were significantly different ($P \leq 0.05$).

[2] Values followed by * were significantly different between years ($P \leq 0.05$).

[3] Species utilized *Maianthemum trifolium, Equisetum sylvaticum* and *Ledum groenlandicum.*

Discussion

Changes in Water Quality

The functional role of wetlands in water quality improvement has been recognized as an essential reason for wetland preservation and creation (Horwitz, 1978). It is therefore predictable that the treatment wetland at Musselwhite Mine influenced the concentration of several of the metals in its discharge effluent. In 1997, Cu and Ni significantly decreased. In 1998, As, Al, and Cu decreased significantly while Fe, Mn and Zn increased significantly (Figure 2). The wetland therefore acted as a sink for some metals and a source for others.

The decrease in Cu was the most noticeable effect. This could be attributed to humic substances which are known to have a high affinity for binding Cu and are present in high concentrations in wetlands (Kadlec and Knight., 1996). Other declines may be attributed to the pH of the wetland system. Aluminum forms insoluble organic, hydroxy or other inorganic complexes at the pH levels present at Sites 1 and 2 (Table 2) (Freda, 1991). Similarly, Ni and As have been retained in wetlands receiving mine effluent that has a pH of 6.8 - 7.0 (Noller et al., 1994). The removal of Ni in surface waters of treatment wetlands has been attributed to wetlands soils, biota or both (Mitsch and Gosselink., 1993). Arsenic mobility is reduced under acidic conditions but increases as pH rises above neutral levels (Foster et al., 1998).

The wetland served as a source for Fe, Mn and Zn. This is somewhat unexpected. A variety of studies have shown that these elements are commonly reduced in wetlands receiving elevated concentrations of these metals (Hammer, 1990, Cooper and B.C. Findlater, 1990, Sobolewski, 1997). The variation in this study may be due to the wetland type and species present. Most studies on wetlands receiving mine effluents deal with created wetlands or at least wetland species adapted to flooded conditions. In this case, many of the species such as reported in Table 1 are not found in continuously flooded conditions, and by the second year of the study had started to decline in abundance. This fact, coupled with the high concentrations of these elements (Table 4) which increase in solubility under anaerobic conditions (Wetzel, 1983), would promote their release into the water column.

However, regardless whether the metals were release or retained in the wetland, the levels which were discharged at the exit of the wetland would still be considered low. The level of 3 μg/L for As (Figure 2) introduced to the receiving environment of Lake 282 is only slightly elevated to the estimated average of As concentration in the world's rivers waters of 2 μg/L (Stephenson, 1987). Aluminum concentrations detected in the treatment wetland are lower than the concentration reported in rivers and lakes which is ten times greater than the 1 mg/L typically found in ocean water (Norseth, 1979). The concentration of Cu detected in the treatment wetland (Figure 2) fall within the reported range found in American rivers waters of 0.83 to 105 μg/L (Durum et al., 1971). Surface water Fe concentrations typically occur at concentrations greater than 200 μg/L and Ni less than 5 μg/L (Hutchinson, 1975) which are well above the levels for these elements in the effluent from the wetland in this study. Surface water Mn concentration is generally less than 100 μg/L (Goldman and Horne., 1983) which corresponds to the findings in this study (Figure 2). Similarly, the Zn concentrations at the wetland exit are lower than the average concentration of 10 μg/L typically found in the world rivers'(Figure 2) (Kadlec and Knight., 1996).

In summary, water quality changed from the entrance to the exit of the wetland. An examination of both the sediment and plant tissue chemistry will determine if any corresponding changes occurred in these components of the wetland.

Changes in Sediment/Soil

In examining the changes in sediment/soil chemistry, it is most appropriate to discuss the changes in concentrations that occurred from the initiation of the wetland flooding in 1997 versus the levels present in 1998. Other studies have shown that it takes two or more years of continuous operation before any trends in wetland functions are evident (Jablonowski, 1995).

Table 4 and Figures 3 - 4 showed that by 1998 the sediment/soil had increased in concentrations of total Cu, Ni, Hg, and Mn and extractable Cu and Ni at Site 1. By Site 3, changes between years for these metals were no longer evident. The increase in sediment/soil concentrations of Cu and Ni between years corresponds to the decrease of these elements in the water column (Figure 2). This is consistent with the tendency for divalent heavy metals such as Cu and Ni to be quickly bound to organic sediments (Schnoor, 1996). It is well known that organic soils bind other heavy metals as well (Otte et al., 1993). In the case of Hg, this is released when terrestrial plant communities are flooded (Moore et al., 1995). In this case, the initial release was apparently countered by re-adsorption of the organic sediment. The increase of Mn in the sediment at Site 1 suggests that the wetland is also reducing the loading of this element. Since the water quality data showed instead that the wetland was acting as a source for Mn, release of this element must have occurred after Site 1. Variation in plant species within the wetland from Site 1 to Site 2 (Table 1) and variations within the wetland for depth and soil are suspected causes of this discrepancy.

Overall, the concentrations of the majority of heavy metals in the sediment/soil were generally present at levels which were within the normal range for natural wetlands or reached normal levels further within the wetland (Figure 3,4, Table 4). The one exception was Mn which had total concentrations considerably higher that the value of 10 ppm for normal unpolluted wetland soils (Kadlec and Knight., 1996). Extractable values for Mn, however, were consistent with those typically found in non-impacted northwestern Ontario sediments (Day and Lee, 1989) and, as noted above, Mn sediment levels quickly declined from Site 1 to Site 2. The Zn concentrations were all below 120 ppm which is normally found in unpolluted wetland soils (Eger, 1993). The Hg concentrations exceeded the normal range of 0.03 - 0.08 ppm (Mirta, 1986) at Sites 1 and 2, but were lowered by Site 3. Total Cu, although high at Site 1, decreased to levels at Site 2 that were typical of the range of 10 - 80 ppm which is normally found in unpolluted sediment (Knezek and Ellis., 1980). The sediment/soil had Mn levels reported higher than 10 ppm which is normally found in unpolluted wetland soils (Table 4 and Figures 2-3) (Kadlec and Knight., 1996). All sites studied at the treatment wetland, with the exception of site 1, exhibited Ni concentrations below the level of 25 ppm which is generally exhibited by wetland sediments (Table 4) (Kadlec and Knight., 1996).

Uptake of Metals by Plants

This study demonstrated the importance of the presence of macrophytes for improving the water quality through the uptake of ions from the sediment/soil that were elevated by the deposition of these elements from the water column. As previously mentioned, the patterns of changes in heavy metal concentrations in created or newly flooded wetlands are not evident after only two years but will become more apparent over time (Jablonowski, 1995). Thus, the 1998 field season provided promising data which resembles the initial stages of a functioning treatment wetland which will improve and mature over time (Tables 5, 6 and Figure 5). The vegetation present in the treatment wetland demonstrated an increase in concentrations of Cu and Ni at Site 1 which corresponds to the elevated concentrations of these elements in the sediment/soil (Table 4). Mercury and Mn which were also elevated in concentration in the sediment at Site 1 but these elements did not have corresponding increases in plant tissue concentrations. Concentration differences existed in the vegetation among sites and these can be attributed to the among-site variation of species causing variation in the uptake of heavy metals (Otte et al., 1993). The concentrations of heavy metals present in the vegetation were detected at levels within the normal range for wetland vegetation (Tables 5,6 and Figure 6). The concentration of Zn present in the vegetation at all sites remained within the non-toxic (for herbivores) range of 25-150 ppm and is considered to be toxic only when present in excess of 500 ppm (Jones, 1972, Walsh and Beaton., 1973). The levels of Fe concentrations detected in this study were considerably lower than 5000 ppm commonly exhibited by aquatic plants and Mn in the vegetation were within the range of 200 - 600 ppm found in wetland plants located on unimpacted sites (Kadlec and Knight., 1996). The Ni concentration detected on unpolluted sites in wetland plants may contain an average of 36 ppm which is higher than the levels detected at all sites in this study (Ohlendorf et al., 1986). The values determined for Cu concentrations in the vegetation were within the normal range of 5 to 20 ppm with the only exception at site 1 (Jones, 1972, Knezek and Ellis., 1980).

Therefore, the plant tissue results suggested that the decreases in metal concentrations in the water column as it passes through the treatment wetland were ameliorated by the effective sediment absorption of heavy metals by the vegetation present. Furthermore, the concentrations of metals that were present in the plants were typical of concentrations found naturally or (in the case of Cu) were quickly dissipated to normal levels further within the wetland.

Effectiveness of the Wetland and Potential Impact on Wildlife

The results showed that the wetland at Musselwhite Mine affected the water quality that is discharged into the receiving water body. Concentrations of some metals were lowered as they passed through the wetland while others increased. However, the rate of removal of metals in such wetlands is a dynamic process based on changes that are still occurring in the wetland. Over time, the performance may change due to alterations in species composition and accumulation of pollutants in the substrate (Berezowsky, 1996). Other factors that will affect the performance of such wetlands are water depth, vegetation density, residence times and flow path length (Martin, 1995). Water velocity is greatly reduced in wetlands, which allows for solids to be entrapped by sedimentation and nutrients to be taken up or transformed by plants and microorganisms (Brix, 1989). If the water velocity in the wetland is increased, the physical heterogeneity and biological diversity will diminish and the self purification capacity will be reduced. As future changes occur in the Musselwhite wetland, it will be essential to continue monitoring the wetland and develop appropriate retention times and flow through velocities so that both factors contribute to optimum performance.

A potential threat to wildlife exists in a treatment wetland ecosystem since it is exposed to toxic compounds (Brix, 1989). The threat arises from toxic compounds in wetlands that may end up in the food chains. But if there is an adequate diversity of wetland plant species present then the animal populations present in the wetland will be less likely to be at risk since they will have a choice for their nutritional and habitat requirements (Kadlec and Knight., 1996). At Musselwhite Mine the diversity of species in the treatment wetland provided a sufficient removal of heavy metals from the effluent that would not adversely affect wildlife populations in general and Moose in particular. For example, many of the wetland plant species present (Table 1) are not a part of moose diets. Even those that are such as *Equisetum* spp., *Salix* spp., and *Nuphar* spp. (Jackson et al., 1991), had metal values lower than the average levels in natural wetlands (Kadlec and Knight., 1996). Therefore, the concern of magnification of metal concentrations as they are passed through the food does not seem to be a problem at this time.

In conclusion, the created wetland at the Musselwhite Mine reduced the concentrations of Cu, Ni, As, and Al discharged to the receiving environment. Concentrations of Fe, Mn, and Zn in the discharge water were slightly elevated by the wetland. These metals were either absorbed or released by the underlying sediment and taken up by the plant species present. Elevated levels of Cu and Ni in plant tissue occurred at the entrance of the wetland but decreased to background levels further within the wetland. None of the metals at the concentrations measured in the plants were thought to be detrimental to herbivores. The wetland is still developing and continued monitoring is required to assess its future performance.

Chapter 2

The Effects of Heavy Metal Uptake at Dona Lake Mine: A Closed Gold Mine Facility

Introduction

One of the vital issues facing the mineral extraction industry is the cleanup of abandoned sites from historic mining and closed mining wastes. The reclamation of the disturbed sites often becomes quite complex due to the presence of acid mine drainage (AMD) and the presence of toxic substances in high concentrations which affects both surface and groundwaters (Ripley et al., 1996). Acid mine drainage describes any undesirable characteristics of mine water even though dissolved metals may be the only issue in a given situation (Barbour, 1994).

Any mineral deposit containing sulphide becomes a potential source of AMD. The most commonly associated minerals associated with AMD are sulfur, copper, zinc, silver, gold, lead and uranium (Gray, 1996). Waste rock, tailings and particulate matter may be chemically reactive and consequently generate soluble acids, soluble metal salts, and radionuclides (Ripley et al., 1996). This causes the formation of several species of soluble hydrous ion sulfates, the production of acidity and the subsequent leaching of metals (Nordstrom, 1982). Total acidity is comprised of mineral acidity, predominately species of iron, along with aluminum, manganese, zinc and other metals varying with the geologic deposit and hydrogen ion acidity (Kent, 1994). Other metals and trace elements in the geologic material may also be solublized due to the acid leaching conditions. Oxidized iron precipitates as ferric hydroxide as the pH increases above 3.5, while pH values of at least 5 and 7 are required to precipitate aluminum and manganese hydroxides, respectively (Kent, 1994).

In order to minimize chemical transformation and erosion, the containment of the waste piles requires isolation from air and weathering. Tailings and other mining wastes, which remain bare and dry, have high susceptibility to wind erosion. Waste rock and the outer walls of tailing impoundment areas can be eroded by rain and snowfall thereby washing sediments onto adjacent land areas and into water bodies (Schnoor, 1996).

The most effective way to control the chemical reactivity of sulfide wastes is to submerge them in the water of a wetland and thereby depriving them of oxygen (Ripley et al., 1996). The use of wetlands has been recognized as a method of improving water quality in nature and are proving to be a reliable and accepted alternative as treatment systems of AMD (Jablonowski, 1995). Over four hundred wetlands have been constructed in the past decade to treat and control the effects of acid mine drainage (Thomson and Turney., 1995). In fact, the examination of natural marshlands surrounding mining areas have led to the advancement of constructed wetlands as a potential treatment option for waters polluted with heavy metals and other toxins (Robinson and Barnes., 1997). The main drawback with the treatment is the reported uptake of high levels of heavy metals into the vegetation and the subsequent release to the environment (Turney and Thomson., 1993).

The release of heavy metals from AMD from the tailings area of a closed mine is the primary concern in this study. The objective was to determine if heavy metal release due to acid mine drainage resulted in corresponding elevated metal concentrations in the impacted vegetation and soil.

Methods

Field Site

The closed Dona Lake Mine site located approximately 10 km south of Pickle Lake, Ontario, Canada (51°42' N 90°6' W) was the focus for this study (Figure 1). Ross-Finlay Ltd. operated the facility with the purpose of extracting gold until August 1994 when production ceased and the mine was acquired by Placer Dome NA Ltd.. Placer Dome NA Ltd. has been reclaiming the tailings area since that time.

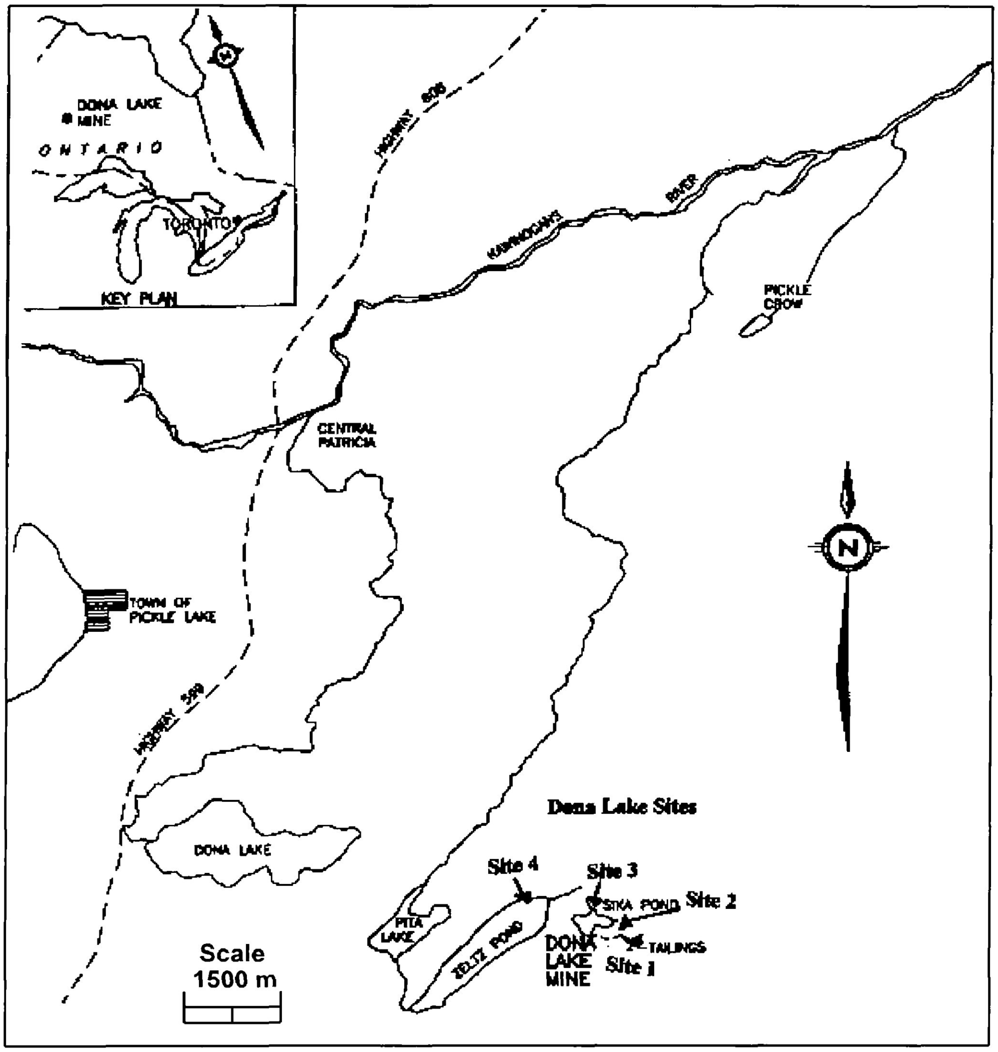
The sampling took place in mid June, July, September 1997 and in early June, July, and August 1998. Four study sites were selected extending from the tailings area to the receiving environment. As the distance increased from the point source, the diversity of plants present also increased. The four sites were: Site 1 - the tailings area, where the original dry tailings were released into the tailing impoundment and are partially submerged, Site 2 - the migrating tailings, located in the area of a small creek immediately behind the tailings dam, Site 3 - the polishing pond, Sika Pond, an open water marsh located downstream from the migrating tailings, and Site 4 - the receiving environment, Zeltz Pond, an open water marsh located downstream from the polishing pond.

Sampling and Laboratory Procedures

At each of the areas, permanent quadrats were established to assess seasonal changes in the vegetation and sediment/soil. Random sampling of the vegetation layers for browse and herbaceous components, and sampling of soil layers was completed for each area. At each area, two vegetation species were sampled, one common to all sites and one specific to each particular site (Table 1). Two sediment/soil samples were collected in the middle of the

Figure 1

Location of the Dona Lake Mine study site locations (a) Site 1, (b) Site 2, (c) Site 3, (d) Site 4.


DONA LAKE MINE
ONTARIO
TORONTO
KEY PLAN
HIGHWAY 808
KAWINOGANS RIVER
PICKLE CROW
CENTRAL PATRICIA
TOWN OF PICKLE LAKE
N
HIGHWAY 599
DONA LAKE
Dona Lake Sites
Site 4
Site 3
SIKA POND
Site 2
TAILINGS
PITA LAKE
ZELTZ POND
DONA LAKE MINE
Site 1
Scale
1500 m

quadrat from the upper 20 cm soil layer. In addition to the two soil samples collected, appropriate Quality Assurance Quality Control (QA/QC) measures were implemented into the sampling procedures for both the sediment/soil and vegetation samples. The sediment/soil and vegetation QA/QC procedures involved obtaining the following: reference sample, split sample, and a duplicate sample. Field measurements of pH and conductivity of the standing water were measured and recorded.

Table 1
Vegetation Sampled at Dona Lake Mine Site in 1997 & 1998

	Collection of Species		
Site	June	July	September/August
Site 1	-*Typha latifolia* -*Calamagrostis canadensis*	-*Typha latifolia* -*Calamagrostis canadensis*	-*Typha latifolia* -*Calamagrostis canadensis*
Site 2	-*Typha latifolia* -*Eleocharis smallii*	-*Typha latifolia* -*Eleocharis smallii*	-*Typha latifolia* -*Eleocharis smallii*
Site 3	-*Typha latifolia* -*Nuphar variegatum*	-*Typha latifolia* -*Nuphar variegatum*	-*Typha latifolia* -*Nuphar variegatum*
Site 4	-*Typha latifolia* -*Potentilla palustris*	-*Typha latifolia* -*Potentilla palustris*	-*Typha latifolia* -*Potentilla palustris*

The depth to the water table or depth of the standing water, photographs and profiles of the site were also recorded. This procedure was completed for each of the three sampling periods for both of the field seasons. The sampling periods were selected to provide coverage of the major changes in growth phenology of the predominantly perennial species. The sampling periods occurred on June 24- July 1, July 22-29, September 16-23, 1997 and June 8-15, July 14-21 and August 25- September 1, 1998.

Vegetation and Sediment/Soil Analysis

The vegetation and sediment/soil samples were packaged and stored on ice in portable coolers for transport to the laboratory and were kept frozen until prepared for analysis. A portion of each of the sediment/soil samples was oven dried at 80 ^{0}C in a preweighed 20 cm^3 crucible to obtain bulk density measurements. All of the vegetation and sediment/soil samples were dried in a drying oven at 50 ^{0}C for approximately 72 hours. The samples were then gently ground and passed through a 2 mm sieve.

The total metals (Al, Cu, Fe, Mn, Ni, Zn) in the vegetation and sediment/soil samples were digested in a H_2SO_4/HNO_3 matrix, diluted with deionized distilled water (DDW) and analyzed by Inductively Coupled Plasma (ICP) spectroscopy. Extractable metals (Al, Cu, Fe, Mn, Ni, Zn) in the sediment/soil samples were determined according to the methods of Lee (1986)(extraction in 0.1M HCl) and analyzed by ICP spectroscopy.

The total cyanide in the vegetation (1.0 g sample) and sediment/soil (2.5 g sample) were extracted for 16 hours in 50 mL 1% NaOH, filtered by gravity with Whatman 1 filter paper, diluted with DDW and filtered through a solid phase extractor and analyzed by a Skalar Autoanalyzer SAN-System. A nitrate correction was implemented to determine the correct total cyanide content in the sediment/soil samples for 1 ppm NO_3 = 20 ppb CN_T. The nitrate in the 0.50 g sediment/soil samples were extracted for 30 minutes with 50 mL of DDW, filtered by gravity with Whatman 1 filter paper decanted into IC vials and analyzed by Ion Chromatography (IC).

The arsenic analysis for the vegetation and sediment/soil samples were digested in H_2SO_4/HNO_3 matrix to convert all forms of As to As (IV). The As in this form was measured by Hydrides Generation Atomic Absorption Spectroscopy at Accurassay Laboratory in Thunder Bay, Ontario Canada.

The mercury analysis for the vegetation and sediment/soil samples were digested in $HCl/HNO_3/KMnO_4$ matrix to extract Hg. After digestion the excess $KMnO_4$ was reduced with hydroxilamine hydrochloride and the concentration of Hg in digest was determined by Cold Vapor Hydrides Generation Atomic Absorption at Accurassay Laboratory.

All of the procedures followed standard operating procedures of LUEL, which were developed from the U.S. Environmental Protection Agency (EPA), Standard Methods of APHA, Environment Canada and the Ontario Ministry of the Environment.

Data Analysis

The computer program SPSS 7.5 standard version statistical package (Windows, 1996) was used in the data analysis. The data were examined for outliers along with presence of skewness and kurtosis. Means of the chemical variables in the sediment/soil samples, and the vegetation species and for *Typha latifolia* were calculated. A two-way analysis of variance (ANOVA) was preformed on the elements in both the sediment/soil and vegetation to assess if there were any significant differences of metals with respect to site and time. A Least Significance Difference (LSD) paired comparison test was preformed on the vegetation and sediment/soil if the ANOVAs were significant among sites to determine which sites differed from each other.

Results

Water Analysis

The pH, conductivity and depth of water measured in the field at the four sites during 1997 and 1998 are shown in Table 2. The pH of the water ranged from 7.05 to 8.87 over the two years (Table 2). For both years, conductivity at site 1 increased in successive months while conductivity levels at sites 3 and 4 remained fairly stable, and site 2 demonstrated a large variation between 370.5 to 812.1 (Table 2). Water depth at sites 3 and 4 was considerably greater in comparison to sites 1 and 2.

Table 2
Properties of Water Reported of Each Site at Dona Lake Mine

		pH			Conductivity (1/ Ω)			Depth (cm)		
Site	Year	June	July	Sept/Aug	June	July	Sept/Aug	June	July	Sept/Aug
1	1997	7.65	7.81	6.52	379.2	453.0	526.5	22.0	17.2	12.2
	1998	7.75	7.52	8.19	433.0	495.1	600.1	10.4	8.0	11.4
2	1997	7.31	7.52	6.98	370.5	636.4	812.1	8.5	13.0	9.6
	1998	7.02	6.82	8.45	510.2	295.7	518.5	15.8	6.4	8.8
3	1997	8.66	7.74	7.18	246.7	272.8	294.4	43.8	64.4	49.6
	1998	8.56	7.48	8.55	257.6	231.6	244.1	74.0	69.2	60.0
4	1997	7.05	7.53	7.05	88.4	96.7	93.3	47.5	43.4	32.0
	1998	8.22	7.34	8.87	98.9	90.1	100.9	51.8	33.0	31.2

[1] depth of standing water is a mean of four corners and centre of quadrate

Sediment/Soil Analysis

Bulk density and paste pH of the sediment/soil at the four sites during 1997 and 1998 are shown in Table 3. Paste pH of the sediment/soil at the four sampling sites ranged from 6.61 to 7.85. Values for pH tended to be lower in 1998 versus 1997. Bulk density of the sediment/soil were similar over the season and between years in each site. Sites 2 and 3 were substantially lower in bulk density in comparison to the remaining sites.

Table 3
Properties of Sediment/Soil of Each Site at Dona Lake Mine

		Paste pH			Bulk Density (g/cm^3)		
Site	Year	June	July	Sept/Aug	June	July	Sept/Aug
1	1997	7.33	6.98	7.51	1.162	1.427	1.354
	1998	7.09	6.81	6.93	1.496	1.419	1.281
2	1997	7.43	7.46	7.51	0.369	0.327	0.340
	1998	6.85	6.61	7.08	0.330	0.345	0.218
3	1997	7.30	7.54	7.43	0.620	0.626	0.742
	1998	7.30	7.17	7.02	0.782	0.578	0.657
4	1997	7.78	7.83	7.85	1.347	0.925	1.143
	1998	6.87	6.97	7.06	1.409	1.086	1.306

[1] bulk density and paste pH are means of the samples collected at each site.

Total and Extractable metal and CN concentrations in the sediment/soil at the four sites during 1997 and 1998 are shown in Table 4 and Figures 2 and 3.

The concentrations of total Al, Fe, and Mn in the sediment/soil present at the Dona Lake Mine site decreased from site 1 - site 4 for both years and for Zn in 1997, and As in 1998. Ni in 1997 and Zn in 1998, and Hg and Cu in both years had the highest concentrations present at site 1 and 3 (Table 4 and Figure 2). A one way ANOVA revealed significant differences among sites for all total metals with the exception of Hg in 1997 and Hg and Cu, in 1998 (Table 4). Paired comparison of means showed that As, Al, Cu, Fe, Mn, Ni, Zn in 1997 and As, Fe and Zn in 1998 were significantly higher at site 1 versus the remaining sites (Table 4). One exception was the concentration of Cu which was significantly higher at site 3 throughout 1997 (Table 4 and Figure 2). Significant differences existed for total As, Hg and Cu with respect to year (Table 4). The Mn concentration decreased from site 1 to site 4 in 1997; concentrations of Al, Fe and Zn in 1997 and 1998 (Table 4 and Figure 3). Cu and Ni concentrations increased for both years and exhibited highest levels at sites 1 and 3 (Table 4 and Figure 3). A one way ANOVA revealed significant differences among sites for all elements concentrations with the exception of Ni in 1998 and CN in both years (Table 4).

Paired comparison of means showed that site 1 was statistically different from the other sites for all element concentrations with the exception of Mn in 1997 and Cu and Zn in 1998 (Table 4). Significant differences existed for extractable Ni and Zn with respect to year (Table 4).

Vegetation Analysis

The mean metal concentrations in vegetation at the four sites during 1997 and 1998 are shown in Table 5 and Figure 4. The metal concentrations illustrated an irregular trend among sites and between years (Figure 4) and a one way ANOVA revealed no significant differences for any of the contaminants among sites with the exception of CN in 1998 (Table 5). CN was the only element to demonstrate a significant difference between years (Table 5).

The total metal concentrations in *Typha latifolia* at the four sites during 1997 and 1998 are shown in Table 6 and Figure 5. A one way ANOVA revealed significant differences among sites for Cu in 1997 and Hg, Al, Fe and CN in 1998 (Table 6). Concentrations for Hg, Al and Fe in 1998 were significant higher in site 1 than the other sites (Table 6 and Figure 5). In 1998, total Al and Fe demonstrated site 1 to be significant different from all of the other sites (Table 6). Total Hg, and Cu all were significant different between years (Table 6).

Table 4
Means of Element Concentrations (ppm) in Sediment/Soil at Dona Lake Mine

		Site Location			
Element	Year	1	2	3	4
As	1997 *	4.55 a	2.43 b	0.838 c	1.05 c
	1998	4.28 a	1.10 b	0.533 b	0.475 b
Hg	1997 *	6.84	4.31	5.73	0.271
	1998	0.009	0.031	0.160	0.009
Total Al	1997	14311.68 a	7554.64 b	6686.78 bc	4983.42 c
	1998	9013.83 a	7509.59 ab	5447.28 bc	4072.63 c
Total Cu	1997 *	91.62 a	58.30 a	225.91 b	11.68 c
	1998	86.48	56.23	79.39	26.20
Total Fe	1997	68294.18 a	55653.36 b	15399.57 c	8860.56 c
	1998	84698.82 a	43463.59 b	15305.83 bc	7870.37 c
Total Mn	1997	662.11 a	421.95 b	221.46 c	148.66 c
	1998	516.03 a	387.81 a	214.80 b	141.33 b
Total Ni	1997	37.37 a	17.84 b	28.36 c	7.17 d
	1998	38.65 a	25.39 ab	20.69 b	13.48 b
Total Zn	1997	136.09 a	74.77 b	28.81 c	17.73 c
	1998	178.28 a	49.07 b	93.88 ab	53.17 ab
Extractable Al	1997	281.51a	112.99 b	73.44 b	37.25 b
	1998	226.02 a	127.06 b	33.05 c	30.69 c
Extractable Cu	1997	9.31 a	0.691 a	19.91 b	0.439 a
	1998	5.85 ab	0.705 a	8.22 b	0.802 a
Extractable Fe	1997	919.93 a	333.06 b	189.53 bc	45.99 c
	1998	1256.31 a	220.29 b	81.88 bc	56.14 c
Extractable Mn	1997	137.86 a	126.55 a	57.84 b	41.82 b
	1998	133.45 a	91.73 b	33.88 c	50.31 c
Extractable Ni	1997 *	1.02 a	0.446 b	2.29 c	0.312 b
	1998	5.21	3.73	5.37	4.68
Extractable Zn	1997 *	9.29 a	6.04 b	2.82 c	0.620 c
	1998	7.54 a	7.14 a	1.26 b	0.627 b
CN	1997	1.09	0.075	0.661	0.044
	1998	5.84	-	0.029	-

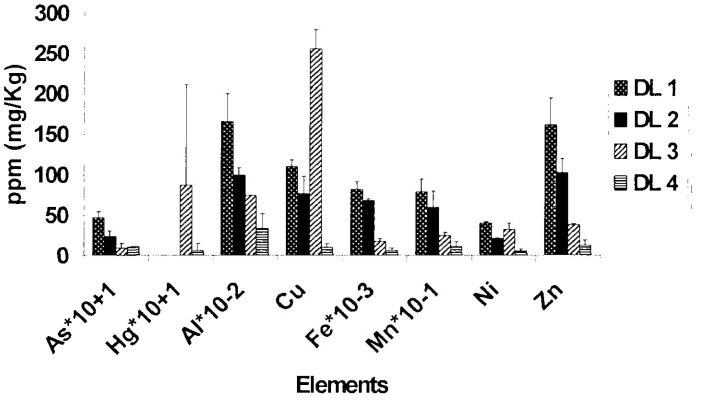
[1] Values followed by different letters in the same year were significantly different (P≤0.05).

[2] Values followed by * were significantly different between years (P≤0.05).

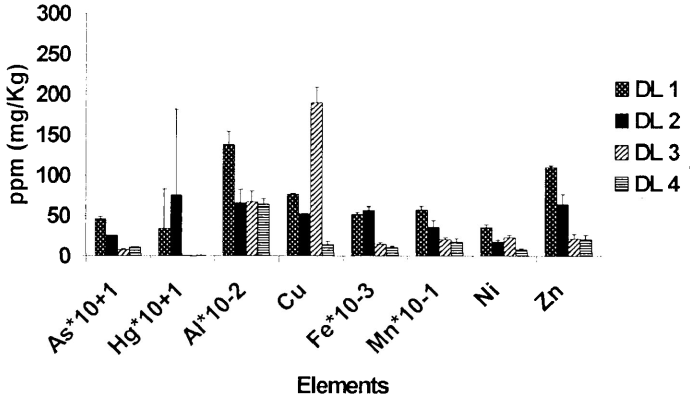
Figure 2

Treatment means of the elements of the Dona Lake sediment: (a) June 1997 total metals in sediment/soil at Dona Lake sites, (b) July 1997 total metals in sediment/soil at Dona Lake sites, (c) September 1997 total metals in sediment/soil at Dona Lake sites, (d) June 1998 total metals in sediment/soil at Dona Lake sites, (e) July 1998 total metals in sediment/soil at Dona Lake sites, (f) August 1998 total metals in sediment/soil at Dona Lake sites.

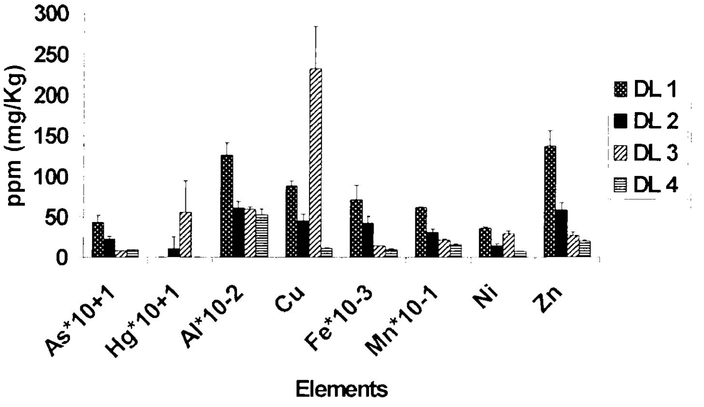
a) June 1997 Total Metals in Sediment/Soil at Dona Lake Sites



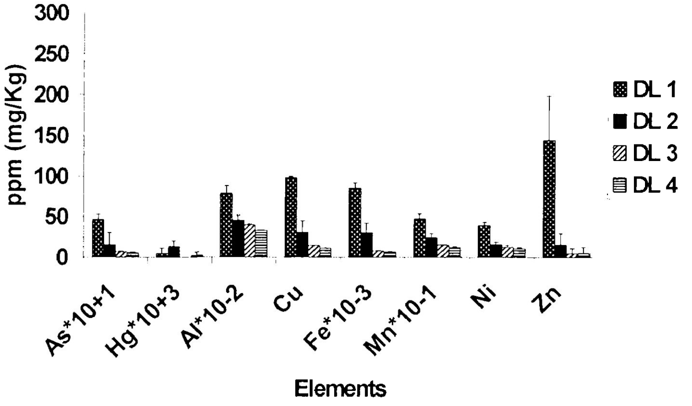
b) July 1997 Total Metals in Sediment/Soil at Dona Lake Sites



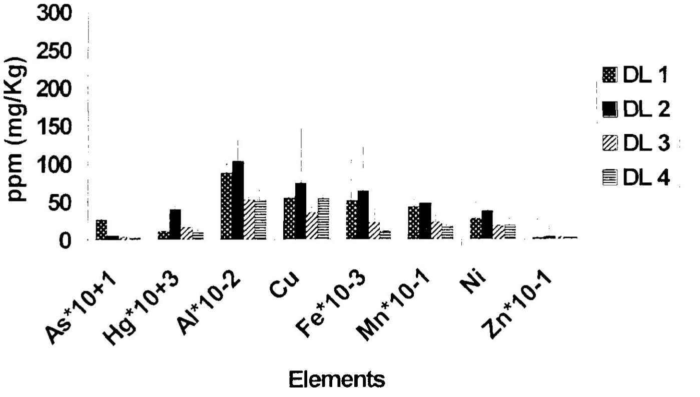
c) September 1997 Total Metals in Sediment/Soil at Dona Lake Sites



d) June 1998 Total metals in Sediment/Soil at Dona Lake Sites



e) July 1998 Total metals in Sediment/Soil at Dona Lake Sites



f) August 1998 Total metals in Sediment/Soil at Dona Lake Sites

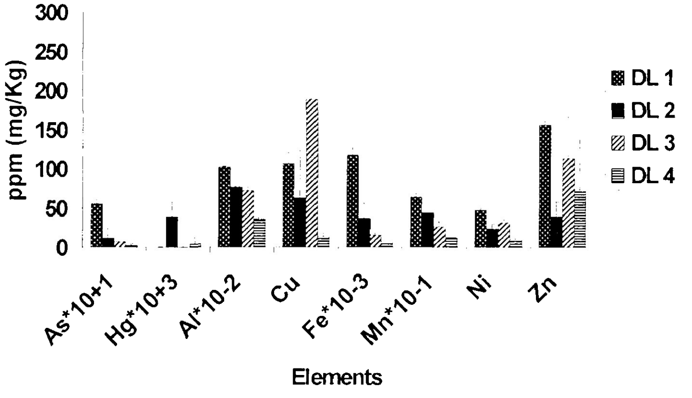
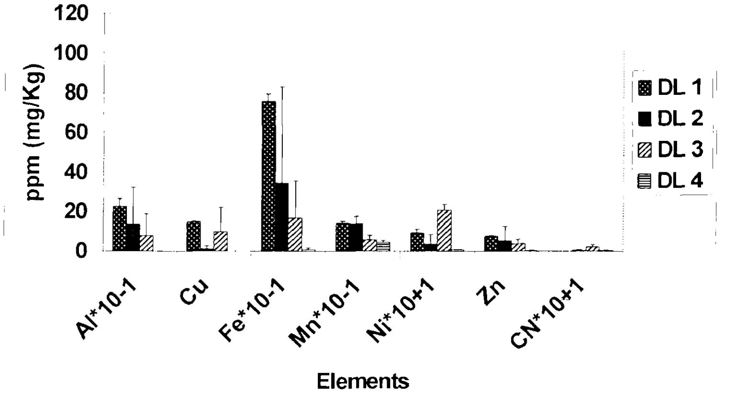
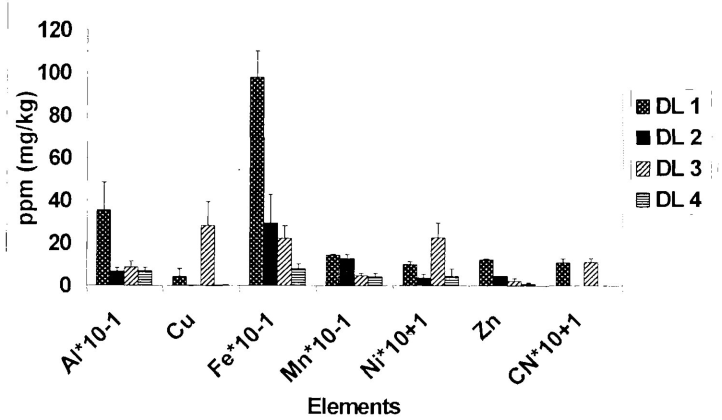
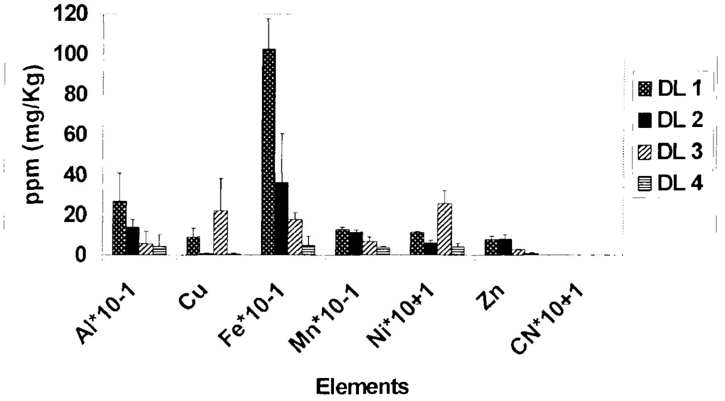


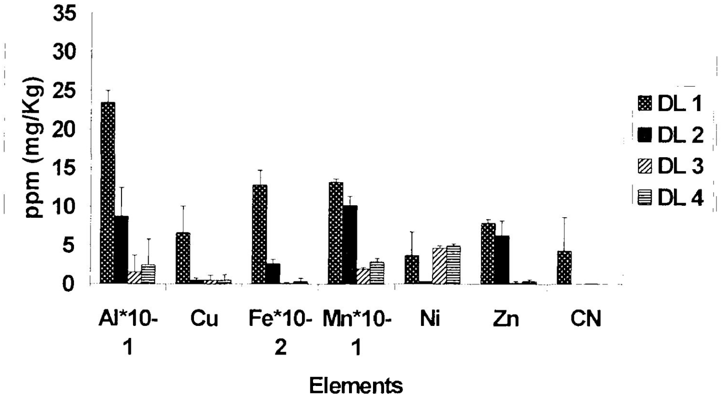
Figure 3

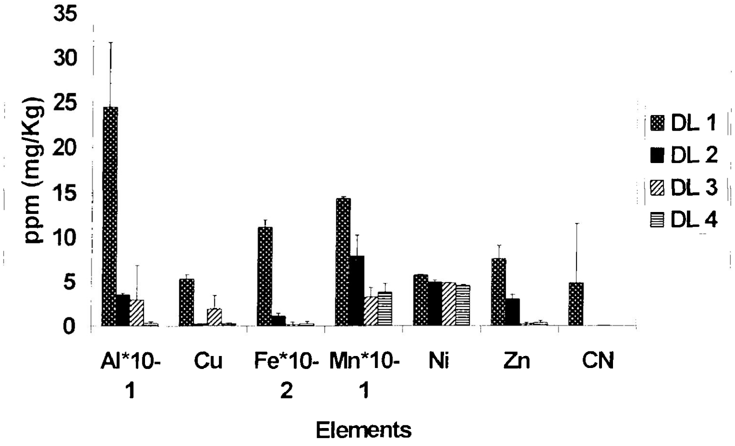
Treatment means of the elements of the Dona Lake sediment: (a) June 1997 extractable metals in sediment/soil at Dona Lake site, (b) July 1997 extractable metals in sediment/soil at Dona Lake site, (c) September 1997 metals in sediment/soil at Dona Lake site, (d) June 1998 extractable metals in sediment/soil at Dona Lake site, (e) July 1998 extractable metals in sediment/soil at Dona Lake site, (f) August 1998 extractable metals in sediment/soil at Dona Lake site.


a) June 1997 Extractable Metals in Sediment/Soil at Dona Lake Sites


b) July 1997 Extractable Metals in Sediment/Soil at Dona Lake Sites


c) September 1997 Extractable Metals in Sediment/Soil at Dona Lake Sites


d) June 1998 Extractable Metals in Sediment/Soil at Dona Lake Sites


e) July 1998 Extractable Metals in Sediment/Soil at Dona Lake Sites

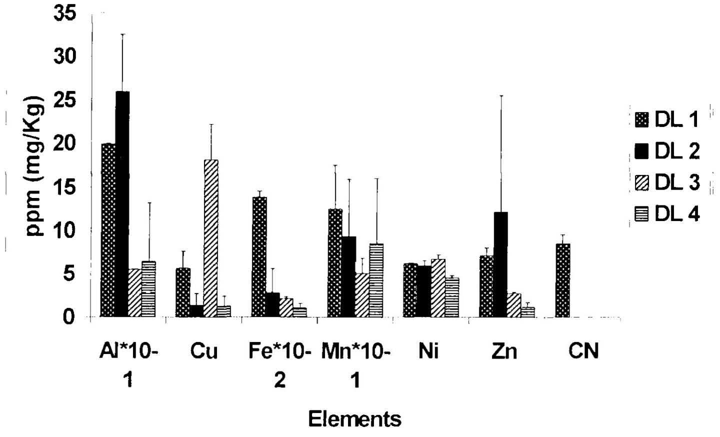
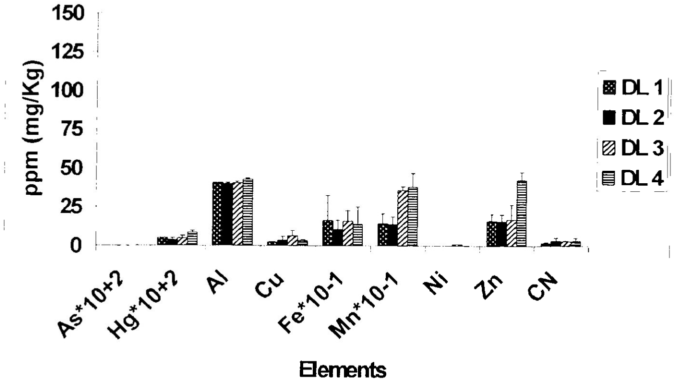
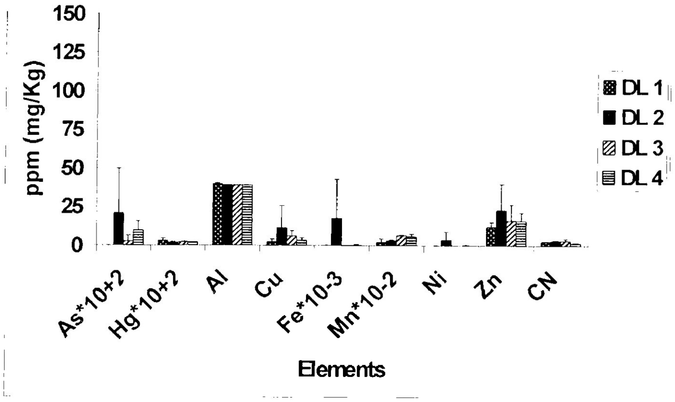
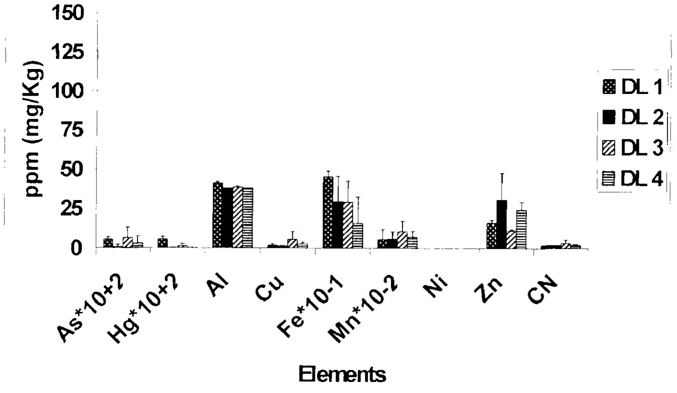

f) August 1998 Extractable Metals in Sediment/Soil at Dona Lake Sites

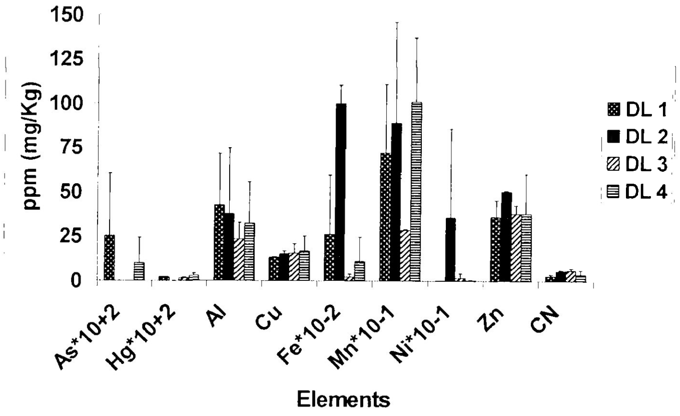
Figure 4

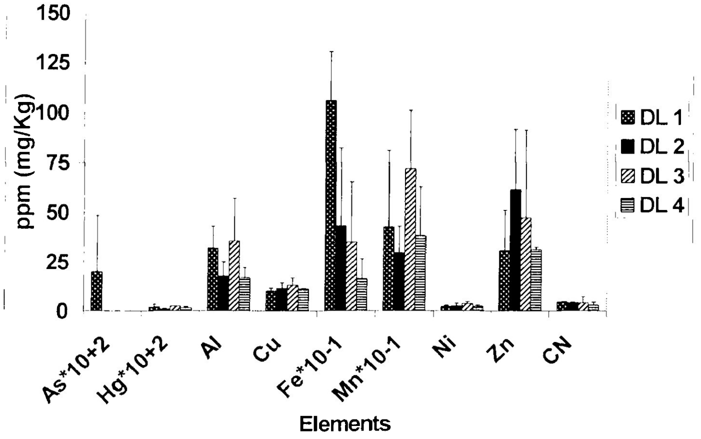
Treatment means of the elements of the Dona Lake vegetation: a) June 1997 total metals in vegetation at Dona Lake sites, (b) July 1997 total metals in vegetation at Dona Lake sites, (c) September 1997 total metals in vegetation at Dona Lake sites, (d) June 1998 total metals in vegetation at Dona Lake sites, (e) July 1998 total metals in vegetation at Dona Lake sites, (f) August 1998 total metals in vegetation at Dona Lake sites.


a) June 1997 Total Metals in Vegetation at Dona Lake Sites


b) July 1997 Total Metals in Vegetation at Dona Lake Sites


c) September 1997 Total Metals in Vegetation at Dona Lake Sites


d) June 1998 Total Metals in Vegetation at Dona Lake Sites


e) July 1998 Total Metals in Vegetation at Dona Lake Sites

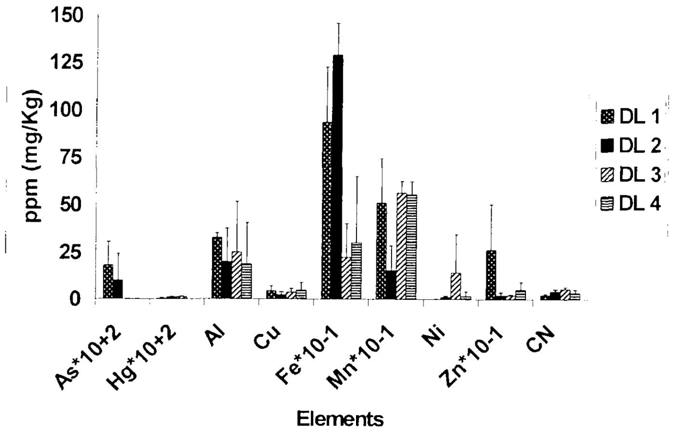

f) August 1998 Total Metals in Vegetation at Dona Lake Sites

Table 5
Means of Element Concentrations (ppm) in the Vegetation at Dona Lake Mine

		Site Location			
Element	Year	1	2	3	4
As	1997	0.054	0.108	0.047	0.065
	1998	0.517	0.100	-	0.100
Hg	1997	0.047	0.019	0.029	0.036
	1998	0.009	0.005	0.012	0.017
Total Al	1997	40.54	38.89	39.46	39.90
	1998	35.66	25.08	28.01	22.64
Total Cu	1997	2.34	5.60	6.16	3.33
	1998	9.23	9.69	10.86	10.99
Total Fe	1997	254.12	6065.43	193.68	232.22
	1998	1531.27	3906.85	264.80	522.56
Total Mn	1997	310.30	353.89	690.26	566.65
	1998	552.16	446.41	523.30	650.13
Total Ni	1997	0.050	3.79	1.02	0.185
	1998	1.29	119.72	11.80	2.67
Total Zn	1997	15.02	23.17	14.97	27.76
	1998	110.14	44.58	35.21	40.23
CN	1997 *	2.16	2.85	3.29	2.50
	1998	3.19 a	4.67 ab	5.22 b	3.38 a

[1] Values followed by different letters in the same year were significantly different ($P \leq 0.05$).

[2] Values followed by * were significantly different between years ($P \leq 0.05$).

Table 6
Means of Element Concentrations (ppm) in *Typha latifolia* at Dona Lake Mine

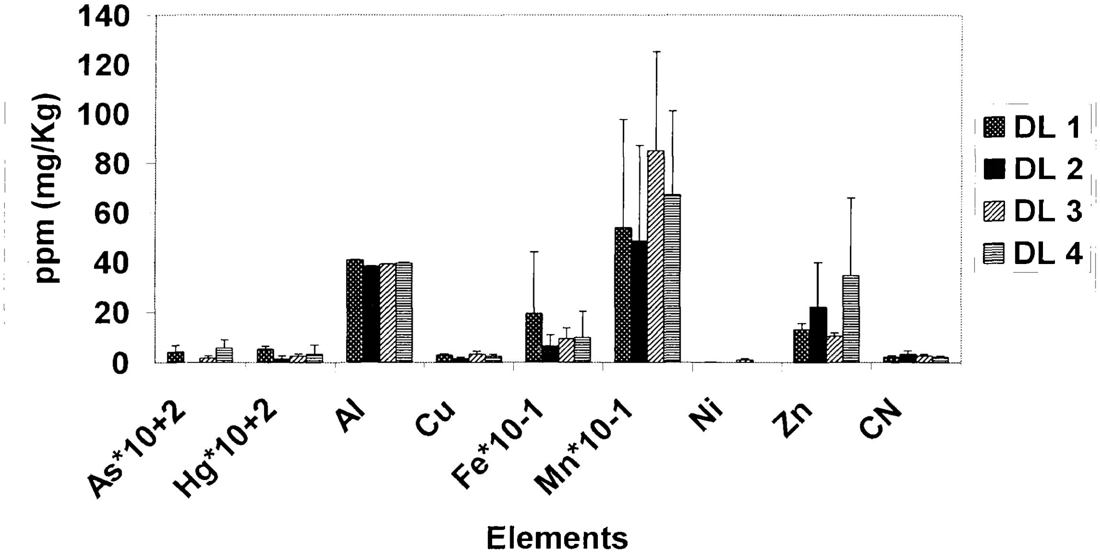
		Site Location			
Element	Year	1	2	3	4
As	1997	0.043	-	0.018	0.058
	1998	-	-	-	-
Hg	1997 *	0.054	0.016	0.026	0.033
	1998	0.016 a	0.006 bc	0.011 ac	0.009 ac
Total Al	1997	40.95	38.65	39.41	39.70
	1998	31.87 a	10.48 b	14.53 b	10.89 b
Total Cu	1997 *	2.98 ab	1.65 a	3.40 b	2.61a
	1998	8.16	8.32	8.23	8.03
Total Fe	1997	193.66	65.88	95.96	100.89
	1998	987.59 a	150.98 b	112.31 b	99.88 b
Total Mn	1997	538.83	485.18	850.36	672.39
	1998	789.78	374.22	467.16	810.03
Total Ni	1997	0.100	-	1.00	-
	1998	1.33	238.30	19.63	3.24
Total Zn	1997	13.02	22.06	10.53	34.70
	1998	159.63	33.89	45.61	24.80
CN	1997	2.27	3.41	2.72	2.27
	1998	3.29 a	4.51ab	5.94 b	4.69ab

[1] Values followed by different letters in the same year were significantly different ($P \leq 0.05$).

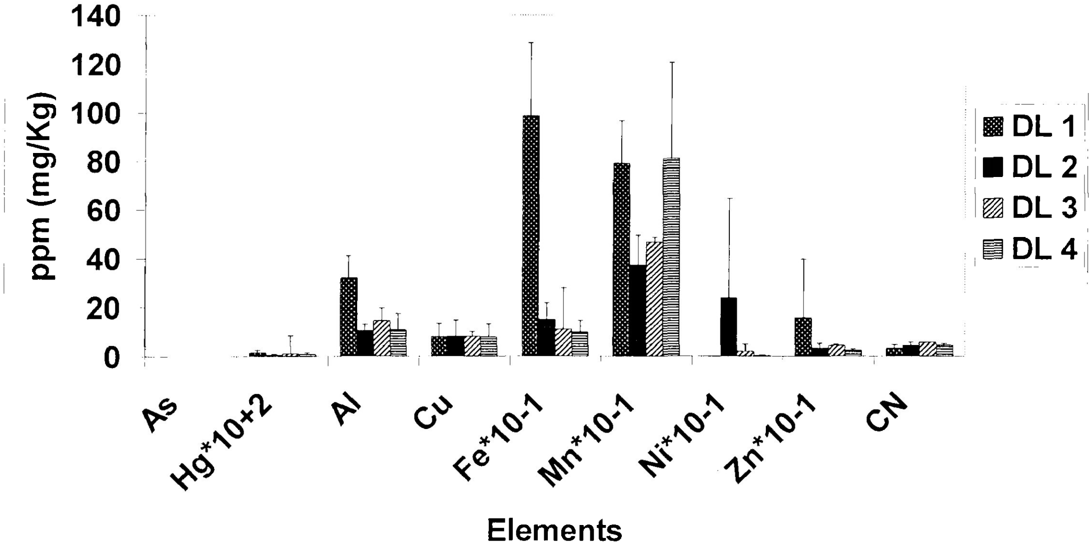
[2] Values followed by * were significantly different between years ($P \leq 0.05$).

Figure 5

Treatment means of the elements of the Dona Lake Typha latifolia species: (a) mean of total metals in Typha latifolia at Dona Lake Mine 1997, (b) mean of total metals in Typha latifolia at Dona Lake Mine 1998.



a) Mean of Total Metals in *Typha latifolia* at Dona Lake Mine 1997



b) Mean of Total Metals in *Typha latifolia* at Dona Lake Mine 1998

Discussion

Acid Drainage

Tables 2 and 3 showed that both the water and the soil at the Dona Lake sites tended to be near or greater than a pH of 7. Acid production from the tailings area at Site 1 was therefore not occurring and increased concentrations of metals due to enhanced solubilities under acidic conditions (Nordstrom, 1982) was not a factor. Nonetheless the results clearly showed that metals were elevated due to leaching from the tailings area and these in turn affected both the soil and vegetation.

Changes in Sediment/Soil

The total and extractable metal concentrations in the sediment/soil at Dona Lake Mine followed similar trends from Site 1 to Site 4. Generally variations of metal concentrations were related to the release from the tailings present at site 1 (Table 4 and Figures 2-3). The significantly higher levels at site 1 gradually decreased with distance from the tailings impoundment. A similar pattern of gradual dilution of metal contamination as distance increases from the point source have been reported in a variety of literature (Bell et al., 1995, Brady, 1990, Barbour, 1994, Down and Stocks, 1977, Filion et al., 1990, Frostman, 1993, Hossner and Hons., 1992, Orava and Swider., 1996, Pierzynski et al., 1994, Ripley et al., 1996). Variations in this pattern did occur for Cu and Hg. For these two metals, levels were higher at Sites 2 and 3 than Site 1. The lower bulk densities at these two sites (Table 3) were indicative of the higher organic content at both sites versus Site 1. Since organic content is known to increase metal complexation (Antonovics et al., 1971, Pierzynski, 1994 #79 , Kadlec, 1996 #14)], the elevated organic content at Sites 2 and 3 was also a key factor in the metal concentrations observed.

As mentioned in the Methods section, as distance increased from the point source, the diversity of plants present also increased. This increased diversity is likely related to the concentrations of metals present in the soil and may be indicative of toxic levels in the upper Sites. Kadlec and Knight (1996) reported that Mn and Ni concentrations in unpolluted sites averaged 10 ppm and 25 ppm respectively. In this study, Mn exceeded 100 ppm at all sites, while Ni was greater than 25 ppm at Site 1. Eger (1993) reported that Zn concentrations in natural wetlands were less than 120 ppm. In this study Zn levels exceeded this concentration at Site 1. Finally Cu concentrations in unpolluted sediment would be expected to fall between 10 ppm - 80 ppm (Knezek and Ellis., 1980). At Dona Lake, total Cu levels exceeded this at both Sites 1 and 3. As explained above, elevated Cu concentrations at Site 1 likely developed by drainage from the oxidizing environment at the tailings area while at site 3, the elevated concentrations of Cu resulted from an increase of sorption capacity due to the presence of organic matter. The total concentrations, however, may not reflect the potential effect on plant growth. Table 4 showed that extractable levels of Cu at Site 1, which are a better indication of plant availability, were nearly as high at Site 1 as Site 3 even though the total concentrations at Site 3 in 1998 was over twice that of Site 1. Copper in particular is strongly bound by carbon compounds (Pierzynski et al., 1994) and Cu tied up in such complexes may only become available for plant growth through time after it is released through decomposition (Brady, 1990).

Uptake of Metals by Plants

In comparing the overall concentrations of the metals in plant tissue at the four sites (Fig. 4, Table 5) little correlation appeared between those sites with elevated sediment metal concentrations in their sediment. This is not unexpected since the overall concentrations were an average of concentrations from different species at the four sites. Although the metal composition in which the plant exists affects the availability for uptake (Down and Stocks, 1977), metal uptake and tolerance is also species specific (Nicolls et al., 1965, Tyler et al., 1989). However, in examining the single species, *Typha latifolia,* there were statistical differences in the uptake of Cu in 1997 and Hg, Al, Fe, and CN in 1998 (Table 6). In most cases, these elevated tissue concentrations could be related to elevated metal concentrations at the same sites. For example, Fe in both years and Al in 1998 declined in the plant tissue as the concentrations of these two metals also declined in the sediment from Site 1 to Site 4. In those cases (Cu and Hg) where the concentrations in the sediment seemed to be related to organic content as well as distance from the tailings, the concentrations of these two metals were highest at Sites 1 and 3. In the case of CN, no clear pattern occurred and suggests that uptake of CN is by a different mechanism than that for metals which typically exhibit luxury consumption (Ricklefs and Miller., 1999).

It is possible that some of the metals in the sediment were in sufficient concentration to be toxic to some plant species. However, it is impossible to isolate which specific metal(s) could be causing the problem. Comparing the levels of metals in the tissue of *Typha latifolia* at the four Dona Lake sites (Table 6 and Figure 5) gives little indication of the specific cause. The Cu levels in *Typha latifolia* were marginally lower at all sites than the 200 ppm reported by Zhang (Zhang et al., 1990) for the roots, shoots and rhizomes. The Zn concentrations for Sites 2-4 were within the range of 10 to 100 ppm determined for plants on unpolluted wetlands (Lan et al., 1990) and even at Site 1 were lower than the 341 ppm at a polluted lead and zinc mine reported by (Lan et al., 1990). Mayer and Gorham (1951) found Mn concentrations of 870 μg/g and 412 μg/g in *Typha latifolia* which corresponds to the levels present in this study. In terms of Ni, the average of 36 ppm found by (Ohlendorf et al., 1986) was higher than the reported concentrations present at Dona Lake with the exception of site 2 in 1998. In fact the concentrations of Ni in *Typha latifolia* at Dona Lake correspond better to those reported by Eger (1993) at an unpolluted site in Minnesota for the leaves (4.9 ppm) and roots (18.0 ppm).

In conclusion, variations in the concentrations of metals occurred in the soil at the closed Dona Lake Gold Mine, but there was no evidence of acid drainage. The concentrations of most metals in the soil declined as distance from the tailings disposal site increased. Increases at some sites were attributed to organic content with a strong sorbing capacity. A single species (*Typha latifolia*) could reflect the differences that existed in sediment concentration of metals. However, when there were a variety of species present with different metal uptake rates, the concentration of metals was not necessarily higher at the most impacted site. Therefore, in assessing the impact of a mine site it is necessary to consider all plant species present and not a single indicator species.

Chapter 3

Vegetative Microcosm Experiments in Gold Mine Tailings

Introduction

Vast areas of abandoned or recently generated mine tailings are common around the world. These unsightly and unproductive tailings left by mining operations present a substantial source of environmental problems leading to air, land and water pollution (Reed et al., 1995). Recent environmental research has focused on the potential to reclaim wastes produced by mining heavy metals (Smith and Bradshaw., 1979).

The tailings from mine waste possess common chemical problems of acidity, salinity, metal content and nutrient deficiencies which primarily restrict reclamation (Hossner and Hons., 1992). This presents a challenge for a successful reclamation project since the presence of high toxic metal concentrations combined with deficient phosphorus, nitrogen and other macronutrients results in the prevention or restriction of plant growth (Antonovics et al., 1971).

The establishment of a successful vegetation cover requires modification of the physical nature of mining wastes. In gold mine tailings, concentrations of metal ions are commonly present in sufficient levels to inhibit or terminate plant growth (Wong, 1986). The presence of heavy metals in tailings leads to a decrease in plant root respiration, water and nutrient uptake, and eventually inhibition of mitosis in root meristems (Gemmell, 1977). The most effective technique to cover the toxic mine tailings was established with a vegetation cover of suitable plants with an adequate inorganic fertilization accompanied by organic amendments (Moore and Zimmermann., 1977).

The reclamation practice in which solid mine wastes are covered by soil and vegetation can reduce wind and water erosion, sulfide oxidation and acid leaching (Bell et al., 1995). However, the most promising approach to prevent acid generation is the practice of covering the acid-producing material with deep water. This creates an anaerobic environment not conducive to oxidation and bacterial action that causes and accelerates the acid-producing reaction (Ripley et al., 1996).

This study examines the potential to reclaim tailings waste through a natural means of creating a wetland environment where the vegetation will aid in the control of heavy metal release and improve appearance. The objective of the study was to determine if a vegetative cover added to gold mine tailings could reduce concentrations of heavy metals and other compounds in the water column.

Methods

Field Site

The Musselwhite Mine, located approximately 130 km north of Pickle Lake, Ontario, Canada (52°37' N 90°24' W) and owned by Placer Dome North America Ltd. was the focus for this study (Figure 1). The Musselwhite gold mine construction began in November, 1995 and it officially opened on July 23, 1997. The open pit and underground establishment was the first mining project in Canada to require review under the Canadian Environmental Assessment Act (CEAA) and the Environmental Effects Monitoring (EEM) program. Several environmental control plans were implemented which included: cyanide recycling within the mill, treatment of tailings slurry using the Inco SO_2/Air process, recycling of tailings pond supernatant to the mill to be used as process water, and directing of pit water, wastewater run off, and site run off, to the polishing ponds for treatment prior to release (Anonymous, 1995). A treatment wetland was established by the expansion of the Snoppy creek, to allow for final treatment with retention times between two to three weeks before release to the receiving environment, Lake 282.

The research experiment was located on the shore adjacent to Crazywind Pond, the tailings impoundment for Musselwhite Mine. A "microcosm" approach was used in this research which allowed for environments of sufficient size to reflect the actual conditions being studied, but small enough to permit treatments to be controlled and monitored. The microcosms in this case were polyethylene containers, 1.8 m in diameter by 1.2 m in height.

Eight containers were used for the experiment in a randomized block design (Figure 2). The design included two of each of the following: (1) control, (2) tailings only, (3) fertilized tailings and (4) organic treatments. The control treatment consisted only of the

Figure 1

Location of Musselwhite Mine.

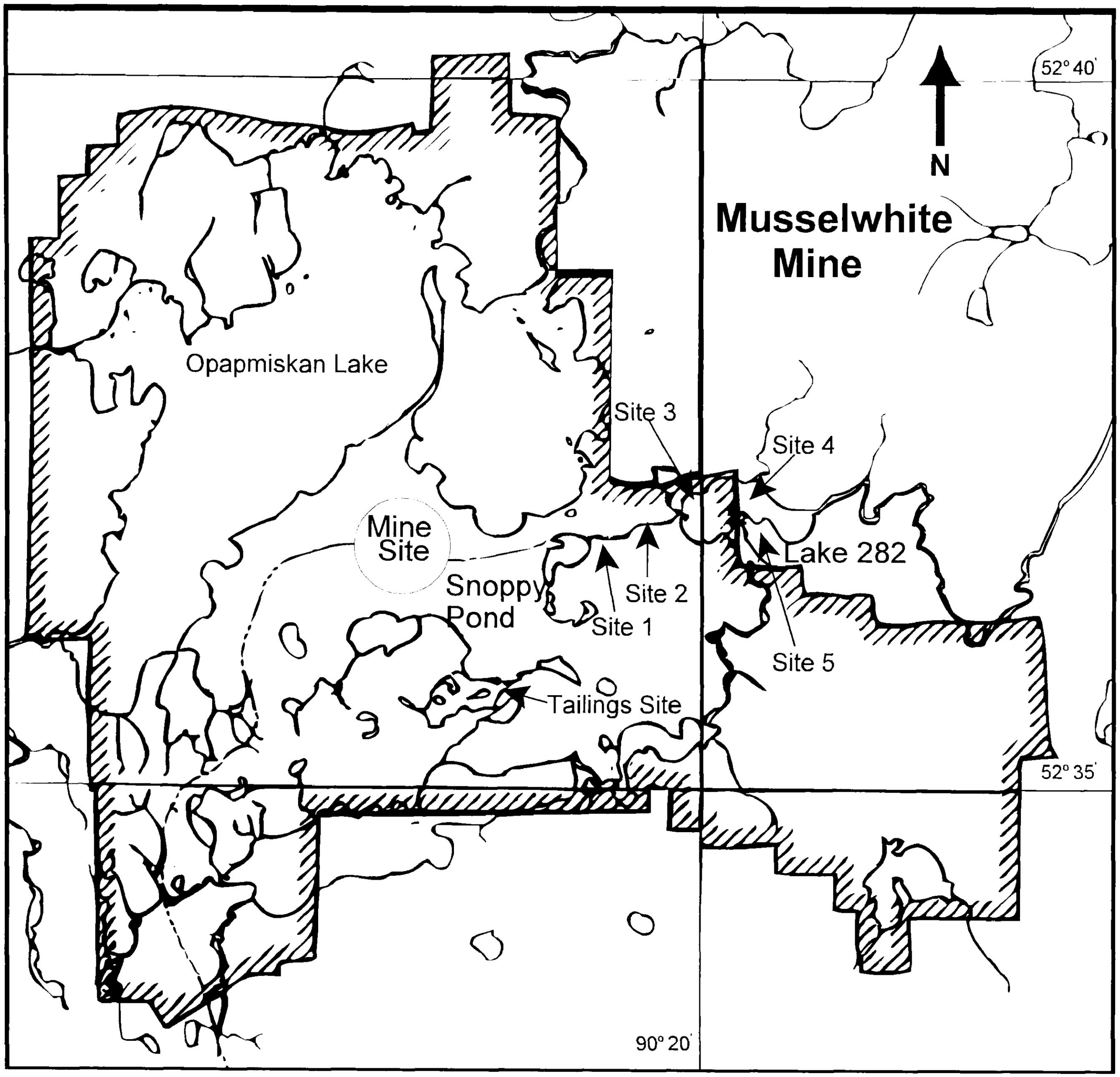

52° 40'
N
Musselwhite Mine
Opapmiskan Lake
Site 3
Site 4
Mine Site
Lake 282
Snoppy Pond
Site 2
Site 1
Site 5
Tailings Site
52° 35'
90° 20'

Figure 2

The microcosm experiment set up.

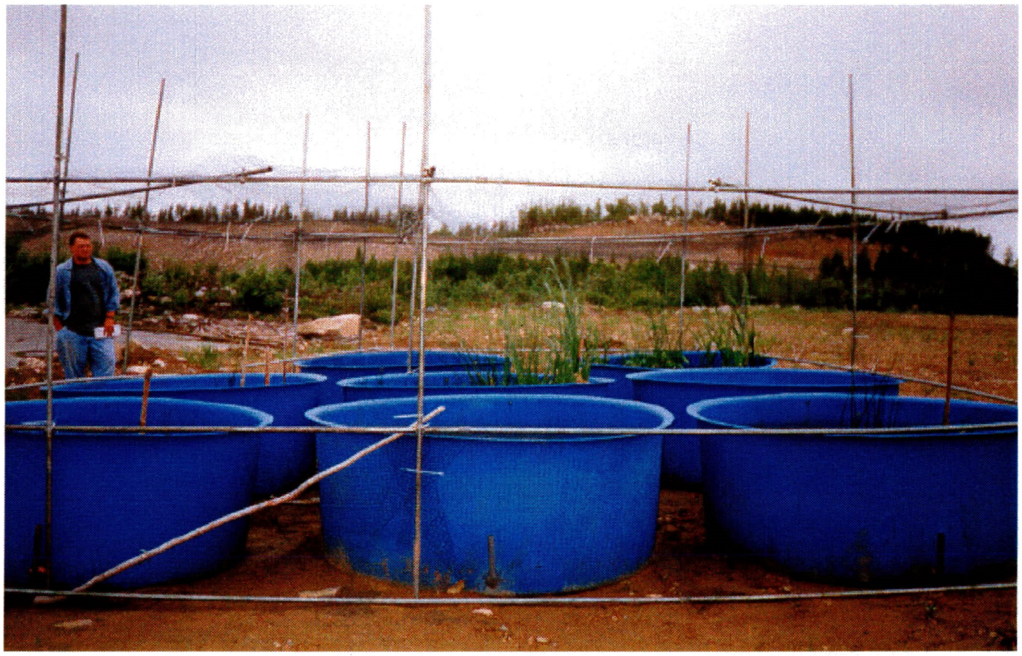


Figure 2 The microcosm experiment set up

tailings and water with no added vegetation. The tailings only treatment included vegetation. The fertilized tailings treatment included the addition of fertilizer pellets and vegetation. The organic treatment included the addition of fertilizer pellets, a 15 cm cap of organic soil and vegetation. The fertilizer used was 20 g pellets of slow release 20-10-5 (N, P, K) manufactured by Sierra Chemical Co. Thirty pellets were added evenly spaced within each bucket.

The eight containers were protected from bird predation by netting. The level of water in the microcosms was maintained with the addition of water from Opapimiskan Lake. The plant species (Table 4) for the experiment were collected at the beginning of the 1998 growing season from Whitefish Lake (48°01' N 90°01' W), located approximately 60 km southwest of Thunder Bay, Ontario, Canada, and then stored and transported to Musselwhite Mine. The plants were transplanted and evenly spaced into each container in the following numbers: 225 mL of *Lemna minor,* 6 of *Nuphar variegatum*, 10 each of *Typha latifolia, Sagittaria rigida*, *Utricularia vulgaris,* and *Myriophyllum verticillatum,* and 75 each of *Scirpus acutus,* and *Eleocharis smallii.* The initial establishment of the microcosms was completed on May 26 – June 1 1998 and the sampling periods occurred on July 14-21 and August 25- September 1 1998.

Sampling and Laboratory Procedures

Following establishment of the microcosms, pH, conductivity and dissolved oxygen were measured in the field during the two sampling periods. Photographs and profiles of the progression of the accumulation of biomass in each microcosm were also recorded. The water, vegetation and tailings were sampled and analyzed at the peak and end of the growing season. An overall seasonal % drop for the total metals present in the water column was calculated. The appropriate Quality Assurance Quality Control (QA/QC) measures were implemented into the sampling procedures for the sediment/soil, water and vegetation samples. The water, sediment/soil and vegetation QA/QC procedures involved obtaining the following: reference sample, split sample, and a duplicate sample. Water collection also included travel and field blanks.

Vegetation and Sediment/Soil Analysis

The vegetation and sediment/soil samples were packaged and stored on ice in portable coolers for transport to the laboratory and were kept frozen until prepared for analysis. A portion of each of the sediment/soil samples were oven dried at 80 ^{0}C in a preweighed 20 cm^3 crucible to obtain bulk density measurements. All of the vegetation and sediment/soil samples were dried in a drying oven at 50 ^{0}C for approximately 72 hours. The samples were then gently ground and passed through a 2 mm sieve. The water samples were preserved with HNO_3 to a pH<2 for the total metals analysis or kept frozen until prepared for the dissolved metals and sulphate analysis.

The total metals (Al, Cu, Fe, Mn, Ni, Zn) in the vegetation and sediment/soil samples were digested in a H_2SO_4/HNO_3 matrix, diluted with deionized distilled water (DDW) and analyzed by Inductively Coupled Plasma (ICP) spectroscopy. The water samples were preserved with 0.5 mL of 1N HNO_3 to a pH<2 and kept at 4°C until analysis. At room temperature the mixed sample was decanted into a 50 mL calibrated quartz test tubes, 0.5 mL of concentrated HNO_3, was added and the samples were digested at 100°C until the samples evaporated below 5 mL. The digested samples were transferred to 10 mL sterile plastic test tubes and each sample was brought up to 5 mL by a careful rinse of the original quartz test tube with a few drops of DDW and analyzed by Inductively Coupled Plasma (ICP) spectroscopy.

Extractable metals (Al, Cu, Fe, Mn, Ni, Zn) in the sediment/soil samples were determined according to the methods of Lee (1986) (extraction in 0.1M HCl) and analyzed by ICP. The dissolved metals (Al, Cu, Fe, Mn, Ni, Zn) in the water samples were decanted at room temperature into 10 mL sterile test tubes and analyzed by Inductively Coupled Plasma (ICP) spectroscopy.

The sulphate in the water samples at room temperature was vacuum filtered through DDW pre-washed Whatman 100 filter papers. The sample was decanted into DDW pre-washed IC vials and analyzed by Ion Chromatography (IC).

The total cyanide in the vegetation (1.0 g sample) and sediment/soil (2.5 g sample) was extracted for 16 hours in 50 mL 1% NaOH, filtered by gravity with Whatman 1 filter paper, diluted with DDW and filtered through a solid phase extractor and analyzed by a Skalar Autoanalyzer SAN-System. A nitrate correction was implemented to determine the correct total cyanide content in the sediment/soil samples for 1 ppm NO_3 = 20 ppb CN_T. The nitrate in the 0.50 g sediment/soil samples were extracted for 30 minutes with 50 mL of DDW, filtered by gravity with Whatman 1 filter paper decanted into IC vials and analyzed by Ion Chromatography (IC).

The arsenic analysis for the vegetation and sediment/soil samples were digested in H_2SO_4/HNO_3 matrix to convert all forms of As to As (IV). The As in this form was measured by Hydrides Generation Atomic Absorption Spectroscopy at Accurassay Laboratory in Thunder Bay, Ontario Canada.

The mercury analysis for the vegetation and sediment/soil samples were digested in $HCl/HNO_3/KMnO_4$ matrix to extract Hg. After digestion the excess $KMnO_4$ was reduced with hydroxilamine hydrochloride and the concentration of Hg in digest was determined by

Cold Vapor Hydrides Generation Atomic Absorption at Accurassay Laboratory.

All of the procedures followed standard operating procedures of LUEL, which were developed from the U.S. Environmental Protection Agency (EPA), Standard Methods of APHA, Environment Canada and the Ontario Ministry of the Environment.

Results

Water Analysis

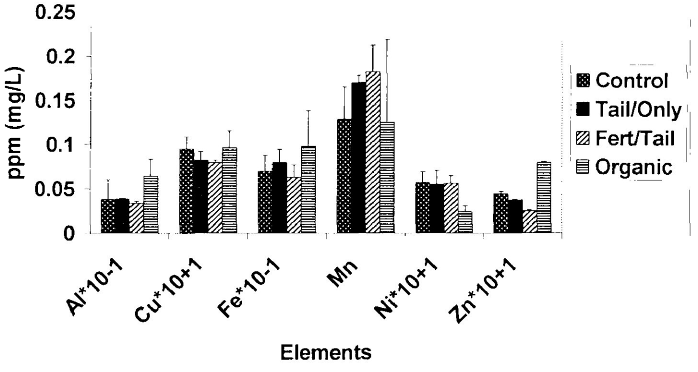
The properties of the water in the microcosms measured in the field during 1998 are shown in Table 1. Dissolved oxygen and pH of the water changed little over the season and were relatively similar among treatments (Table 1). The conductivity levels in June remained consistent in all of the treatments with the exception of the organic treatment which was significantly lower. The treatments with vegetation illustrated a decrease in conductivity levels over the season which is demonstrated by significantly lower levels in the organic treatments (Table 1).

Table 1
Properties of Water Reported of Each Microcosm Treatment at Musselwhite Mine

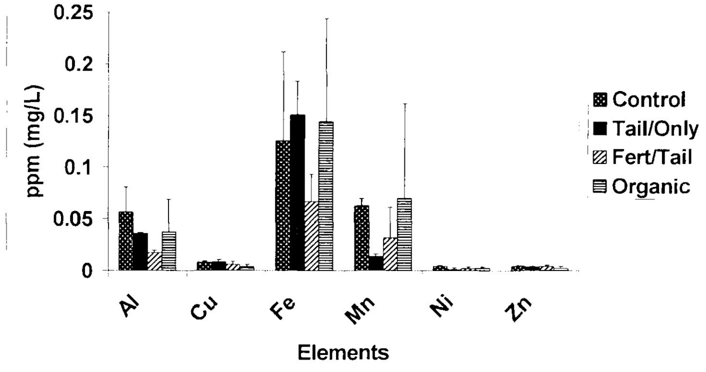
	pH			Conductivity (1/ Ω)			Dissolved O_2 (mg/L)		
Site	June	July	August	June	July	August	June	July	August
Control	7.80	7.29	7.66	1240	1390	1019	N/A	7.24	7.36
Tail/Only	7.70	7.34	8.09	1409	2052	947	N/A	6.26	7.07
Fert/Tail	7.72	7.47	7.76	1361	2023	870	N/A	6.41	6.86
Organic	8.16	7.35	7.37	842	1774	781	N/A	8.01	7.86

[1] pH, conductivity and dissolved O_2 are means of the samples collected in each microcosm.
[2] Dissolved O_2 values obtained at a temperature of 14°C and 15°C for July and August, respectively.

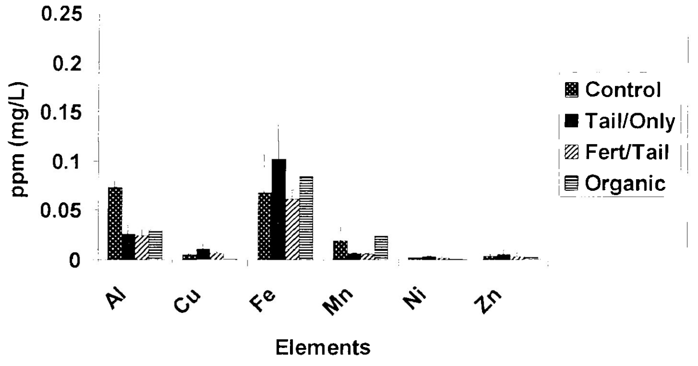
Total metal concentrations in the water in the microcosm treatments sites are shown in Figure 3. The SO_4, total and dissolved metals present in the water of all microcosm treatments initially were equal in concentration for each element among treatments (Figure 3). At the end of the growing season there was a considerable difference in concentration among treatments for all of the concentration levels of SO_4, total and dissolved metals (Al, Cu, Fe, Mn, Ni, Zn) (Figure 3). Compared to the other treatments at the end of the growing season the control and tailings



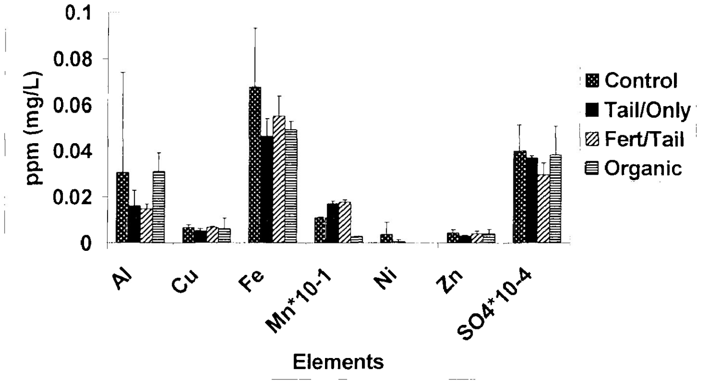
a) June 1998 Total Metals in Water of Microcosm Treatments



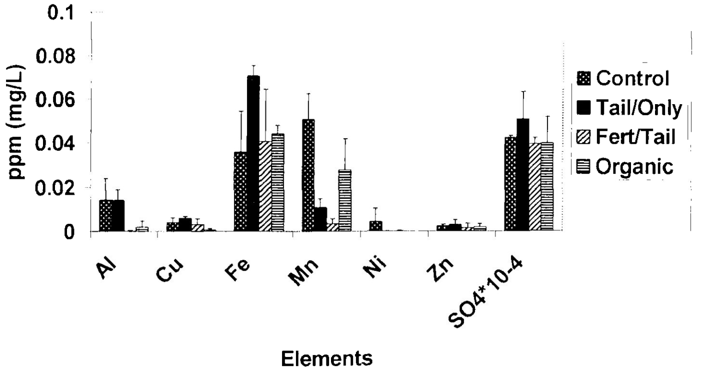
b) July 1998 Total Metals in Water of Microcosm Treatments



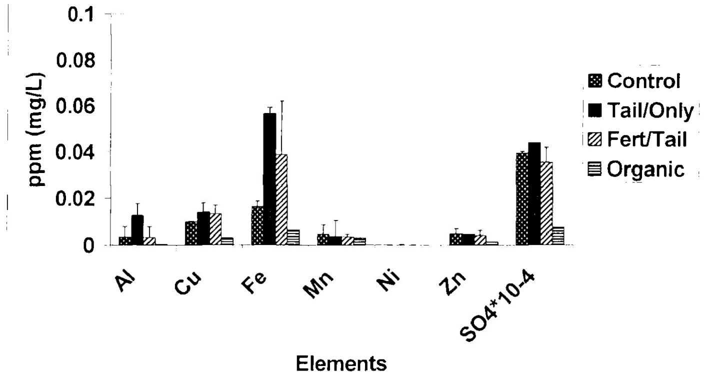
c) August 1998 Total Metals in Water of Microcosm Treatments



d) June 1998 Dissolved Metals in Water of Microcosm Treatment



e) July 1998 Dissolved Metals in Water of Microcosm Treatment



f) August 1998 Dissolved Metals in Water of Microcosm Treatment only treatments exhibited higher SO_4, total and dissolved metals (Al, Cu, Fe, Mn, Ni, Zn) concentrations (Figure 3). The organic treatments at the end of the growing season had the lowest concentration for SO_4, dissolved metals; Al, Cu, Fe, Mn, Ni, and Zn with the majority of total metals; Cu, Ni, and Zn (Figure 3). Aluminum and Fe, in the organic treatments were significantly lower than the highest concentrations which was present in the control and tailings treatments, respectively (Figure 3).

The overall seasonal percentage drop in chemical properties of microcosm treatments are shown in Table 2. Over the entire growing season, the highest percentage drop for SO_4, and total metals were demonstrated in the organic treatments (Table 2). The organic treatment had lower Cu and Zn concentrations (90.5% and 72.5%, respectively) compared to the other elements (Table 2). A significant increase in concentration of Cu and Zn occurred in the control, tailings only and fertilized tailings treatments (Table 2). Total Zn and Ni concentrations were lower in comparsion to the remaining elements but remained higher in comparison to the other treatments. Levels of SO_4 diminished by 80% in organic treatments. Decrease of SO_4 in the control and remaining treatments were less significant (Table 2). Total Al, Fe and Mn decreased relatively the same across the treatments (Table 2).

Table 2
Overall Seasonal % Drop in Chemical Properties Of Each Microcosm Treatment at Musselwhite Mine

		Site			
Element	Total	Control	Tailings/Only	Fert/Tailing	Organic
Al	Total	80.3	93.3	92.8	95.4
Cu	Total	37.9	-23.6	18.5	90.5
Fe	Total	90.2	87.2	90.3	91.5
Mn	Total	84.7	96.0	96.7	80.9
Ni	Total	61.2	45.5	64.3	76.5
Zn	Total	15.5	-53.8	-33.6	72.5
SO_4		2.01	-18.3	-20.4	80.2

Sediment/Soil Analysis

The properties of the sediment/soil in the microcosm treatments during 1998 are shown in Table 3. The paste pH of the sediment/soil varied from 7.0 to 8.1 (Table 3). The bulk density of the sediment/soil was similar among treatments with the exception of the organic treatments which had significantly lower (approximately one third lower) values (Table 3).

Table 3
Properties of Sediment/Soil of Each Microcosm Treatment at Musselwhite Mine

	Paste pH		Bulk Density (g/cm^3)	
Site	July	August	June	August
Control	7.7	8.0	1.632	1.691
Tailing/Only	7.7	7.9	1.412	1.782
Fert/Tail	7.6	7.9	1.448	1.499
Organic	72	7.1	0.647	0.739

[1] bulk density and paste pH are means of the samples collected in each microcosm treatment.

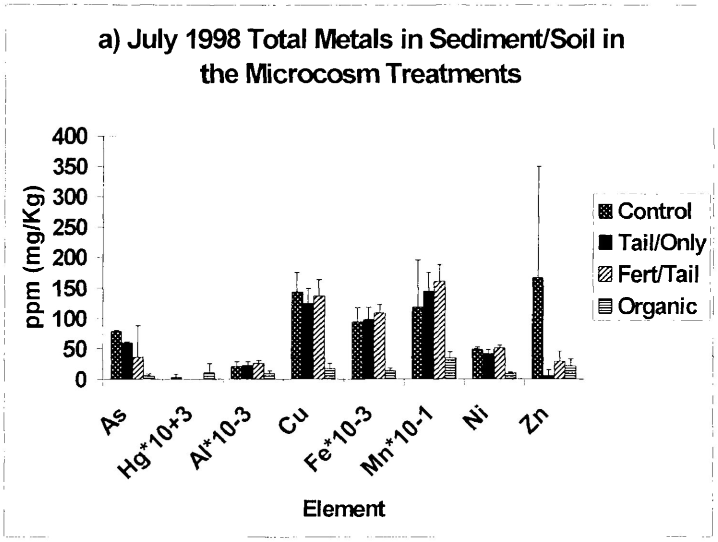
Total metal concentrations in the sediment/soil in the microcosm treatments sites during 1998 are shown in Figure 4. Total concentrations of As, Al, Cu, Fe, Mn, and Ni in the control, tailings only and fertilized tailings treatments were higher in concentration in these treatments than in the organic treatment in both July and August. The organic treatment had the lowest total As, Al, Cu, Fe, Mn and Ni concentrations for both July and August (Figure 4).

Extractable metal concentrations in the sediment/soil in the microcosm treatments sites during 1998 are shown in Figure 4. The control, tailings only, and fertilized tailings treatments had higher concentrations of CN and extractable Al, Cu, and Fe than the organic treatment for both July and August. Cyanide and extractable metals Al, Cu, and Fe were present in the lowest concentration in the organic treatment for July and August (Figure 4). Extractable Ni concentrations present in all of the treatments were relatively the same in concentration for both sampling periods (Figure 4).

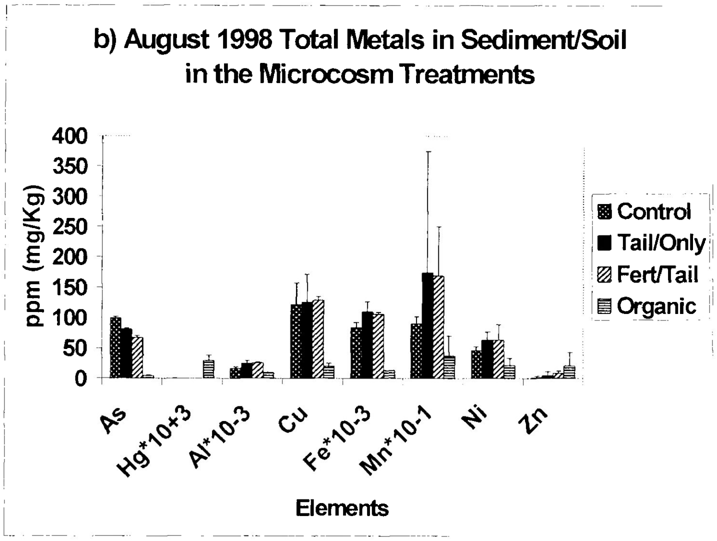
Vegetation Analysis

In general, the vegetation grew poorly in the tailings only treatment; *Nuphar variegatum* and *Typha latifolia* had the highest potential in this treatment with a sparse presence of *Scirpus acutus* and *Sagittaria rigida.* The foliage of the species that survived in the tailings only and fertilized tailings treatment were light green to yellow in most cases. The fertilized tailings treatments had an estimated higher density of plants in comparison to the tailings only treatment. The organic treatment provided an environment in which the transplanted species were able to survive and thrive in the conditions furthermore, species growing in organic treatments propagated the entire living space (Figure 5 and Table 4).

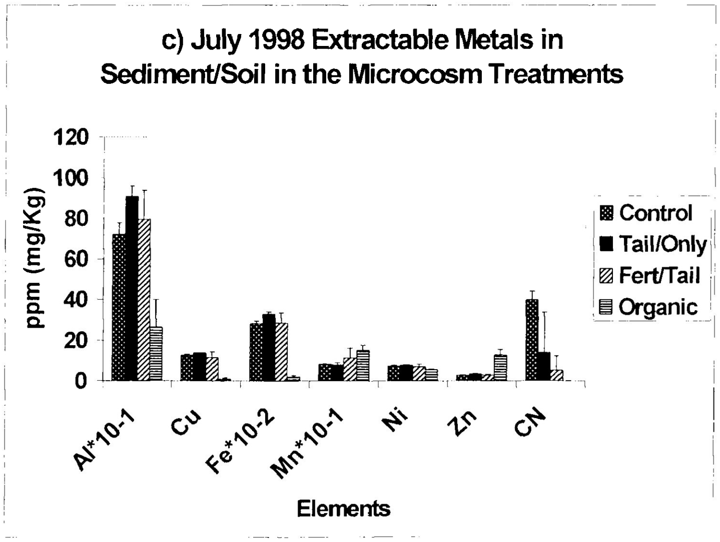
Total metal and CN concentrations in the vegetation in the microcosm treatments sites during 1998 are shown in Figure 6. Uptake varied by treatment, growth form and species. The organic treatment had the lowest uptake of Cu and CN in all species. In the submerged species, *Utricularia vulgaris*, the organic treatment had the lowest concentrations of Fe, Al, Ni, and Zn. The role of the organic cap was less clear in the floating leaf species and in the three emergent species. In *Nuphar variegatum*, it had the highest concentrations for Al, Fe, and Mn. In the emergent species, Fe and Zn were reduced in *Typha latifolia* and *Scripus acutus* but were in the highest concentration in *Sagittaria rigida.* Manganese, on the other hand, was in the highest concentration in the organic treatment for *Typha latifolia* and *Scripus acutus* but reduced for *Sagittaria rigida.* Overall, lowest uptake of metals and CN occurred for *Nuphar variegatum.* The submerged species, *Utricularia vulgaris* contained the highest concentrations of Fe and Al. Of the three emergent species, *Sagittaria rigida* had the highest concentrations of Al, Cu and Fe.



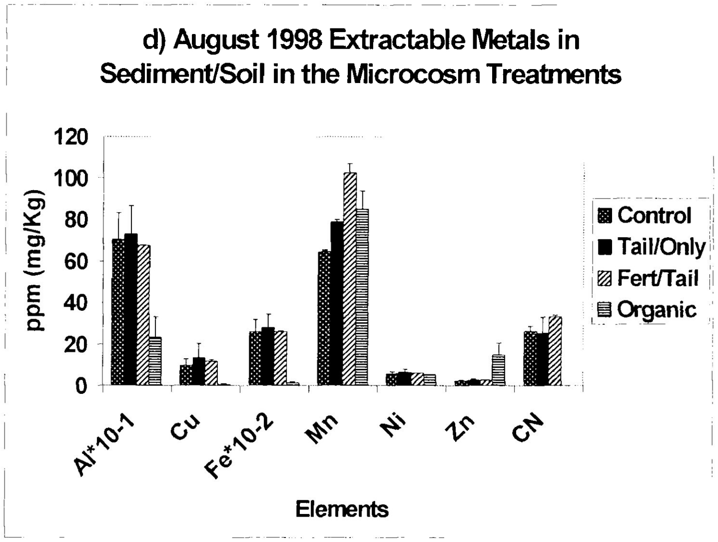
a) July 1998 Total Metals in Sediment/Soil in the Microcosm Treatments



b) August 1998 Total Metals in Sediment/Soil in the Microcosm Treatments



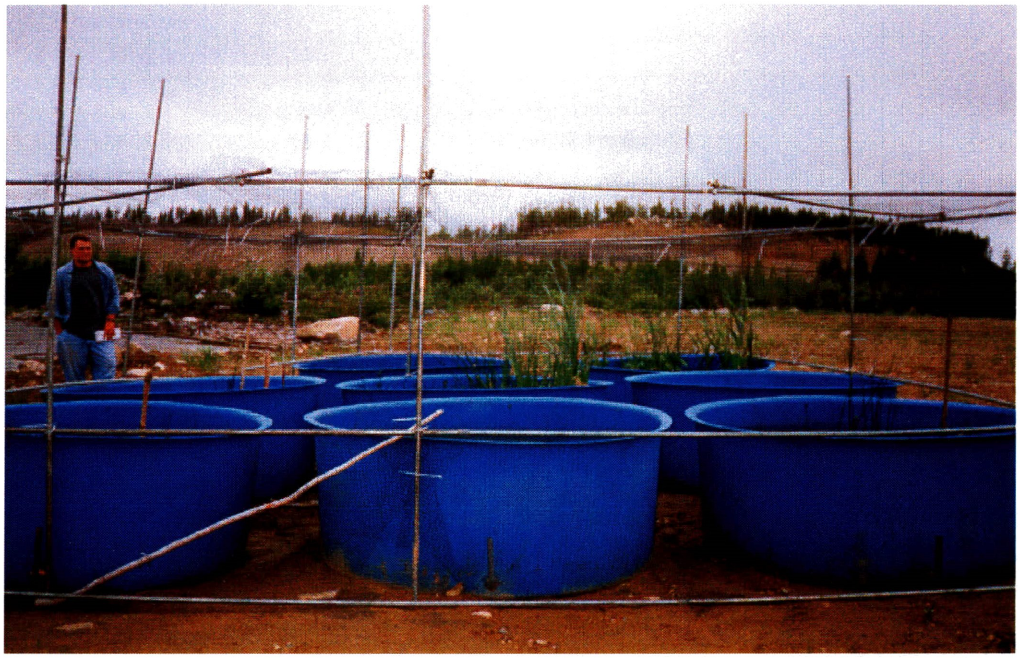
c) July 1998 Extractable Metals in Sediment/Soil in the Microcosm Treatments



d) August 1998 Extractable Metals in Sediment/Soil in the Microcosm Treatments

Figure 5

The progression of the microcosm experiment; (a) the initial stage of the microcosm treatments set up, (b) the organic treatments at the initial stage of the microcosm experiment, (c) the final stage of the microcosm treatment experiment, (d) the organic treatments at the final stage of the microcosm experiment.



(a) the initial stage of the microcosm treatments set up

(b) the organic treatments at the initial stage of the microcosm experiment

(c) the final stage of the microcosm treatment experiment

(d) the organic treatments at the final stage of the microcosm experiment

Table 4
Vegetation and Planted Sampled in the Microcosm Treatments at Musselwhite Mine

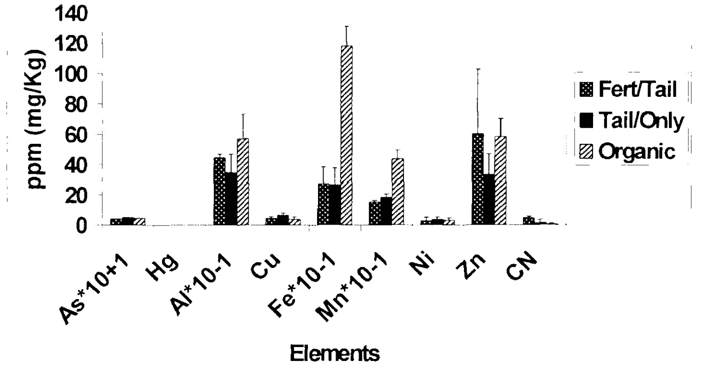
Microcosm Treatment	May Species Transplanted	July Species Extant Sampled (*)	August Species Extant Sampled (*)
Control 1	-No Species Transplanted	-No Species Transplanted	-No Species Transplanted
Control 2	-No Species Transplanted	-No Species Transplanted	-No Species Transplanted
Tailings Only 1	-*Typha latifolia* -*Scirpus acutus* -*Nuphar variegatum* -*Eleocharis smallii* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Lemna minor* -*Sagittaria rigida*	-* *Nuphar variegatum* -*Typha latifolia* -*Scirpus acutus* -*Sagittaria rigida* -*Myriophyllum verticullatum* -*Utricularia vulgaris*	-* *Nuphar variegatum* -* *Scirpus acutus* -* *Utricularia vulgaris* -* *Myriophyllum verticillatum* -* *Typha latifolia* -*Sagittaria rigida*
Tailings Only 2	-*Typha latifolia* -*Scirpus acutus* -*Nuphar variegatum* -*Eleocharis smallii* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Lemna minor* -*Sagittaria rigida*	-* *Typha latifolia* -* *Scirpus acutus* -* *Utricularia vulgaris* -*Nuphar variegatum* -*Sagittaria rigida*	-* *Nuphar variegatum* -* *Scirpus acutus* -* *Utricularia vulgaris* -* *Typha latifolia* -* *Sagittaria rigida*
Fert/Tailings 1	-*Typha latifolia* -*Scirpus acutus* -*Nuphar variegatum* -*Eleocharis smallii* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Lemna minor* -*Sagittaria rigida*	-* *Scirpus acutus* -* *Nuphar variegatum* -* *Utricularia vulgaris* -*Typha latifolia* -*Myriophyllum verticillatum*	-* *Scirpus acutus* -* *Utricularia vulgaris* -* *Myriophyllum verticillatum* -* *Typha latifolia* -* *Nuphar variegatum*
Fert/Tailings 2	-*Typha latifolia* -*Scirpus acutus* -*Nuphar variegatum* -*Eleocharis smallii* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Lemna minor* -*Sagittaria rigida*	-* *Nuphar variegatum* -*Typha latifolia* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Sagittaria rigida* -*Scirpus acutus*	-* *Nuphar variegatum* -* *Scirpus acutus* -* *Typha latifolia* -* *Utricularia vulgaris* -* *Sagittaria rigida*
Organic 1	-*Typha latifolia* -*Scirpus acutus* -*Nuphar variegatum* -*Eleocharis smallii* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Lemna minor* -*Sagittaria rigida*	-* *Typha latifolia* -* *Scirpus acutus* -* *Nuphar variegatum* -* *Lemna minor* -* *Eleocharis smallii* -* *Sagittaria rigida* -*Utricularia vulgaris* -*Myriophyllum verticillatum*	-* *Nuphar variegatum* -* *Scirpus acutus* -* *Utricularia vulgaris* -* *Lemna minor* -* *Typha latifolia* -* *Eleocharis smallii* -* *Sagittaria rigida* -*Myriophyllum verticillatum*
Organic 2	-*Typha latifolia* -*Scirpus acutus* -*Nuphar variegatum* -*Eleocharis smallii* -*Utricularia vulgaris* -*Myriophyllum verticillatum* -*Lemna minor* -*Sagittaria rigida*	-* *Typha latifolia* -* *Scirpus acutus* -* *Nuphar variegatum* -* *Lemna minor* -* *Eleocharis smallii* -* *Sagittaria rigida* -*Utricularia vulgaris* -*Myriophyllum verticillatum*	-* *Nuphar variegatum* -* *Scirpus acutus* -* *Typha latifolia* -* *Sagittaria rigida* -* *Eleocharis smallii*

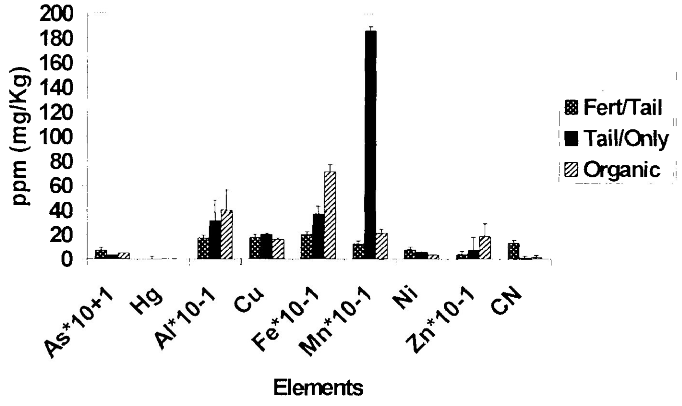
[1] Species marked with * indicates sampled species

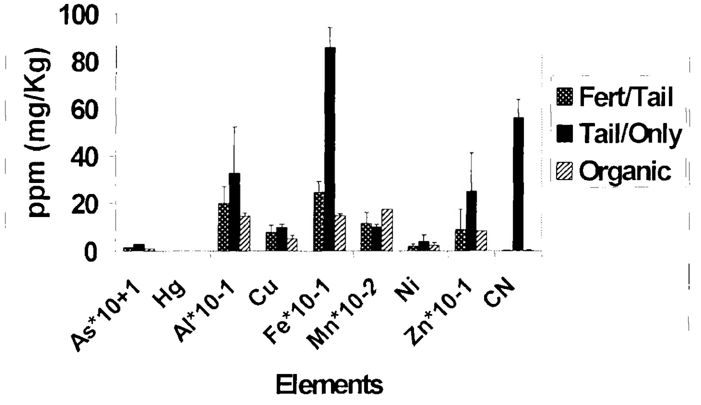
[2] Due to the very low water depth in Aug. the Organic 2 treatment lost the submergent species

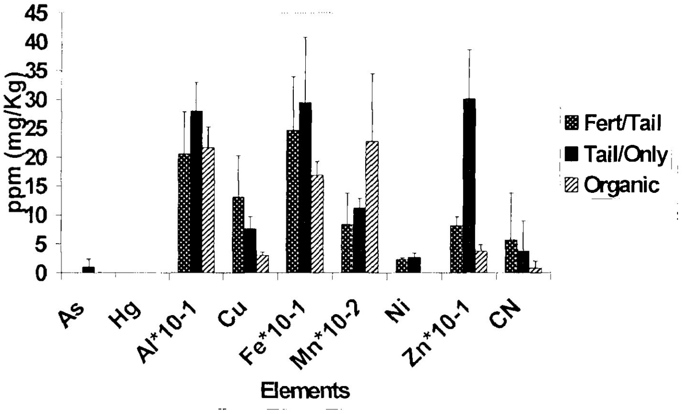
Figure 6

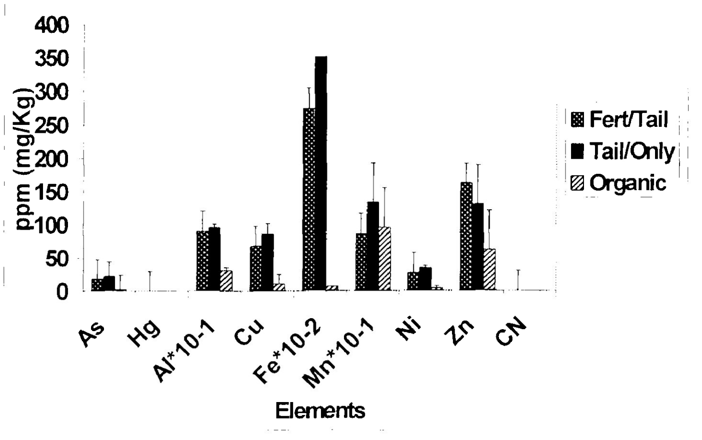
Treatment means of the elements of the microcosm vegetation species: (a) the total metals present in *Nuphar variegatum* in the microcosm treatments, (b)the total metals present in *Sagittaria rigida* in the microcosm treatments, (c) the total metals present in *Scirpus acutus* in the microcosm treatments,(d) the total metals present in *Typha latifolia* in the microcosm treatments, (e) the total metals present in *Utricularia vulgaris* in the microcosm treatments.


a) Total Metals Present in *Nuphar variegatum* in the Microcosm Treatments


b) Total Metals Present in *Sagittaria rigida* in the Microcosm Treatments


c) Total Metals Present in *Scirpus acutus* in the Microcosm Treatments


d) Total Metals Present in *Typha latifolia* in the Microcosm Treatments


e) Total Metals Present in *Utricularia vulgaris* in the Microcosm Treatments

Discussion

Changes in Water Quality

Initially, the concentrations of SO_4 and total metals (Al, Cu, Fe, Mn, Ni, Zn) in the water of each microcosm treatment were relatively equal (Figure 3). However, by the end of the growing season the concentration of all these components varied among treatments (Figure 3 and Table 2). There was a general reduction in metal concentrations in all treatments with the control and tailings only treatments exhibiting the least change. Overall, the organic treatments displayed the lowest concentration and the highest % drop in concentrations for SO_4 and Cu, Ni, and Zn. In the case of SO_4, levels were diminished by 80% in the organic treatment in comparison to a small decrease present in the control and increases in the remaining treatments (Table 2). The changes to these parameters were due to adsorption and release by the tailings treatments.

All the microcosms had above neutral pH and oxidized conditions in their water columns (Table 1) and these conditions would have major influences on the metals studied. Although the metals differ in their solubilities, the pH and toxic levels would encourage precipitation of metals as oxides and hydroxides (Hutchinson, 1975) resulting in the observed metal declines in the microcosms (Table 2). Additional reactions would be expected in the organic treatment and these account for the noticeable declines in Cu, Zn and SO_4 in this treatment versus the others. Organic matter is known to strongly adsorb Cu and Zn (Pierzynski et al., 1994) and the organic compounds present in the water column of this treatment presumably precipitated out with these metals. The organic matter in the bottom of the microcosms, coupled with the occurrence of high concentrations of SO_4, would also promote the growth of sulfur reducing bacteria. These bacteria use SO_4 as an energy source to breakdown the organic matter and are known to accumulate in lakes enriched with sulfate drainage from mine sites (Pedersen et al., 1997). The observed declines in SO_4 would suggest that it had been reduced to insoluble sulfides which would precipitate additional metals (Pedersen et al., 1993) and possibly account for the increased metal depletion in this treatment.

Changes in Sediment/Soil

The declines in the metal concentrations in the water column did not results in corresponding increases in the sediment metal concentrations (Figure 4). Concentrations of both total and extractable metals were within one standard deviation of each other at the start versus the end of the growing season. This is not unexpected. The concentration changes in the water column versus the concentrations of the metals in the sediment are minute. In other studies using microcosms, it has been reported that the experiments have to run until the system has matured before any measurable changes in sediment concentrations will occur (Jewell, 1994). In terms of the metal concentrations present, the organic treatment had lower concentrations of Ni, Fe, Cu, Al, and As compared to the other treatments showing that the organic layer acted as an effective barrier to the dispersion upwards from the underlying tailings in this treatment.

Cyanide, on the other hand, did increase in the tailings only and the fertilizer treatments to levels similar to those found in the control. The organic treatment had no detectable cyanide concentration in July or August. This finding is consistent with the known sorption of CN onto organic detritus which has been found to reduce levels of this contaminant in other wetlands (Higgs et al., 1992).

Uptake of Metals by Plants

There are quantitative differences which exist among plant species in ion uptake and within species requirements for different nutrients (Fitter and Hay., 1981). The differences among plant species in ion uptake in this study were apparent in the uptake of metals by each of the different species in the treatments (Figure 6). Of the five species examined, *Nuphar variegatum* had the lowest concentrations of metals (and CN). However, although there were variations due to species, uptake of several metals by the plants also reflected the relative concentrations of metals present in the sediment for most contaminants. For example, the organic treatment had lower concentrations of CN, Ni, Cu and As in the sediment (Figure 6) and these substances in turn were lower in the plant tissue of all species grown in this treatment. Exceptions to this rule occurred for Fe and Al which followed this pattern exactly (correlation of tissue to sediment concentration) only in *Utricularia vulgaris*. In the other species, either there was no difference among treatments or, in the case of *Nuphar variegatum* and *Sagittaria rigida*, Fe was actually higher in the organic treatments. In these cases, the higher Fe concentrations may reflect the general vigor of the plants. Figure 5 shows that in addition to the much increased growth of all species in the organic treatment, the plants tend to be dark green with no chlorotic patterns. Since Fe is a vital component of the chlorophyll molecule, the lower concentrations of Fe in some of the species may represent a stress caused by excessive levels of other contaminants and a decline in the production of chlorophyll and Fe demand (Devlin and Witham, 1983).

A major concern of the growth of wetland species on tailings areas is that the species will contain abnormally high concentrations of metals which in turn will be passed on through the food chain (Scott, 1972). The results suggest that if an organic cap is added, this should not be a problem for most metals. For example, Hutchinson (1975) reported that normal foliar Ni concentrations were approximately 3 μg/g. Although the concentrations of Ni were greater than this for *Utricularia vulgaris* and *Sagittaria rigida* in the tailings only and fertilized tailings treatments (Figure 6), they were well below this level in the organic treatment. Similarly the normal level for foliar Cu of 48 μg/g (Hutchinson, 1975) was exceeded in the tailings only treatment for *Sagittaria rigida* but was within normal ranges for the organic treatment. Normal levels for Zn in unpolluted wetlands range from 10 - 100 μg/g (Lan et al., 1990), and for Mn, from 200 - 600 μg/g (Kadlec and Knight., 1996). Only *Nuphar variegatum* exhibited values within these ranges for all treatments. The higher levels for both Zn and Mn in the organic treatment relative to the other treatments likely reflects the high levels that were present in the organic sediment (Figure 6). In the case of the other metals, the organic sediment tended to have the lowest values.

Based on the results of this study, it is also interesting to consider which species would be the most desirable for bioremediation from the viewpoint that the best species for this purpose would have the lowest concentrations of metals. As mentioned previously, *Nuphar variegatum* had the overall lowest uptake of metals but the surface area it was able to colonize in the microcosms was limited (Figure 5). The submerged species, *Utricularia vulgaris*, had the highest levels of Fe and Mn in addition to high levels of Zn in the tailings and fertilized tailings treatments, but the organic treatment was able to reduce these concentrations appreciably. Of the three emergent species, *Sagittaria rigida* had the highest uptake of Al, Cu and Fe. *Typha latifolia* had comparable levels to *Scirpus acutus* except for Mn which was considerably higher in *Typha.* This suggests that although *Typha latifolia* has been and reported as a species with an inherent or fundamental tolerance for heavy metals (Antonovics, 1975, McNaughton et al., 1974), other emergent species could be equally as effective.

In conclusion, it was demonstrated that a vegetation cover can be established on gold mine tailings if suitable treatments are used. The most promising treatment in this study was an organic soil cap in combination with added fertilizer. The species considered most suitable in terms of survival and limited heavy metal uptake was *Nuphar variegatum*. Since, the experiment was short in duration a long term outcome of such a cover is not known. Future experiments should examine whether water quality would continue to improve with such a vegetative cover and which particular species are the best for both survival and growth in mine effluent situations over a long period of time.

General Conclusions

The use of a created wetland at the Musselwhite Mine was shown to be a potential alternative natural treatment system for water purification. The wetland was able to reduce concentrations of Cu, Ni, As and Al discharged to the receiving environment. The discharge water contained moderately elevated concentrations of Fe, Mn and Zn. The wetland therefore acted as a sink or source for metals absorption or release from the underlying sediment and uptake by the plant species present. The concentrations of the metals in the plant species were not sufficiently elevated to be of concern for herbivores. In order to maintain the wetland and assess its future performance, it must be monitored during its initial development stages.

The chemical composition of the tailings and the environmental fate and bioavailability of contaminants interacted to affect the surrounding ecosystem at the closed Dona Lake Mine. The release of heavy metals from the tailings area elevated metal concentrations the receiving sediment/soil. This resulted in a reduced biodiversity of plant species at the tailings site which increased as the distance from the tailings disposal area increased. When all plant species were considered at the sampling sites, overall metal concentrations in the plant tissue were similar at all sites. However, when only a single species (*Typha latifolia*) was considered, the metal concentrations in the plant tissue increased as the concentration of metals in the soil/sediment increased.

Reclamation through the establishment of a vegetation cover can be accomplished on gold mine tailings in suitable treatment environments. The treatment with the most effective vegetation cover in this study was an organic soil cap in combination with added fertilizer. *Nuphar variegatum* was found to be the most suitable species with respect to survival and limited heavy metal uptake. Therefore, the reclamation of tailings at Musselwhite Mine by creating a wetland environment where the vegetation will aid in the control of heavy metal release and improve appearance could be feasible. However, it should be noted that further study should be conducted to examine whether water quality would continue to improve and which particular species are the best for both survival and growth in mine effluent situations over a longer duration of time.

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