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8 Impact of Mining Activities on the Terrestrial and Aquatic Environment with Emphasis on Mitigation and Remedial Measures

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Abstract

Mining activities have both local and regional impacts on terrestrial and aquatic ecosystems. Mines produce large quantities of waste-rock and tailings which must be disposed of on land or into aquatic ecosystems. The major results in terms of contamination by heavy metals are areas of wasteland and sources of acid and metal-rich runoff from land-sited tailings piles or waste-rock heaps and the subsequent pollution of soils, lakes, rivers, and coastal areas. After the fact remediation or control of leachates from tailings and waste-rock on land or remediation of waterways polluted by mine tailings are being found to be complicated and expensive. New mines often successfully incorporate mitigation measures that are economically sound in the long-term. This paper overviews these problems and concerns with examples from around the world.

1 Introduction

Mining has gone on for thousands of years and abandoned mines and mining operations are found throughout the world. Mines are local phenomena and their major impacts are largely local but not always and dispersion of heavy metals can be regional, and even global (Allan, 1988).

Today, the main metals mined other than iron and chromium, are copper, zinc, lead and nickel (Table 1). Tonnage produced of all the other metals mined is relatively insignificant in comparison to these six. For example, in 1985, some 1500 tonnes of Au were produced versus some 8,000,000 tonnes of Cu; and some 19,000 tonnes of Cd versus some 6,000,000 tonnes of Zn. The order of metal production in 1985 was Fe > Cr > Cu > Zn > Pb > Ni > Sn > Mo > Co > Cd > Ag > Hg > Au and this has been similar for several decades. In the last few decades, the major producers for the major metals, other than Fe and Cr, have been the U.S.A. for Cu; Canada for Zn and Ni; and the U.S.S.R. for Pb.

Table 1. Historical changes in global production of metals commonly associated with pollution from mining activities (x1000 tonnes). (Abstracted from Nriagu, 1979; Nriagu & Pacyna, 1987.)

Metal	Pre-1850	1850-1900	1900-1940	1950	1960	1970	1980
Copper	45,000	13,000	49,000	2650	4212	6026	7660
Zinc	50,000	15,000	40,000	1970	3286	5469	5220
Lead	55,000	25,000	51,000	1670	2378	3395	3096
Nickel	-	200	1,500	144	327	639	759
Cadmium	-	-	-	6.0	11	17	15
Silver	-	-	-	6.2	7.5	9.4	11
Mercury	-	-	-	4.9	8.3	9.8	7.1
Gold	-	-	-	1.0	1.4	1.5	1.2

Mining and metal benefication industries have developed into sophisticated operations, yet the basic causes of pollution remain unchanged. In a typical metal mine operation (Fig. 1), more water by weight is employed in production of the metal concentrate that the weight of ore grade material itself. Water losses and water gains at a typical metal mine show that the main direct release of metals to freshwaters is from tailings and polishing ponds and emissions later in the benefication stage (Fig. 2).

The objective of the following is to examine the sources of acid mine drainage and related or other heavy metal contamination of the terrestrial and aquatic environment by mining activities; and mitigation or remediation measures to reduce such contamination of terrestrial and particularly aquatic environments.

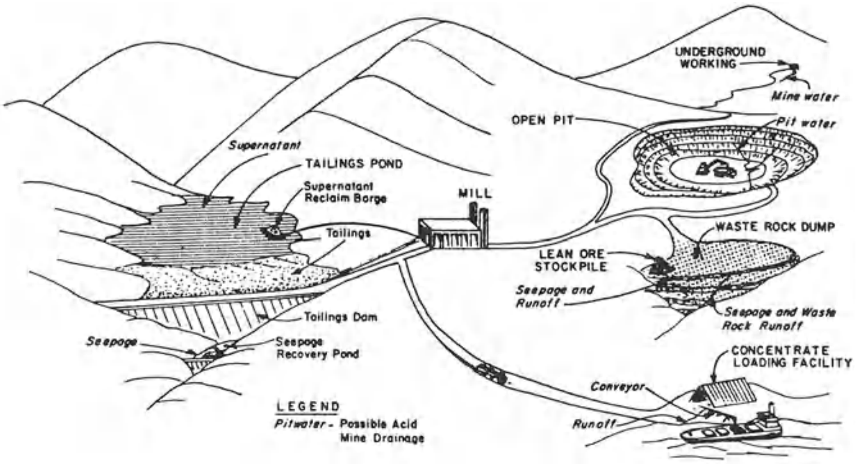

Fig. 1. A typical mining operation. (After Ferguson & Erickson, 1988)

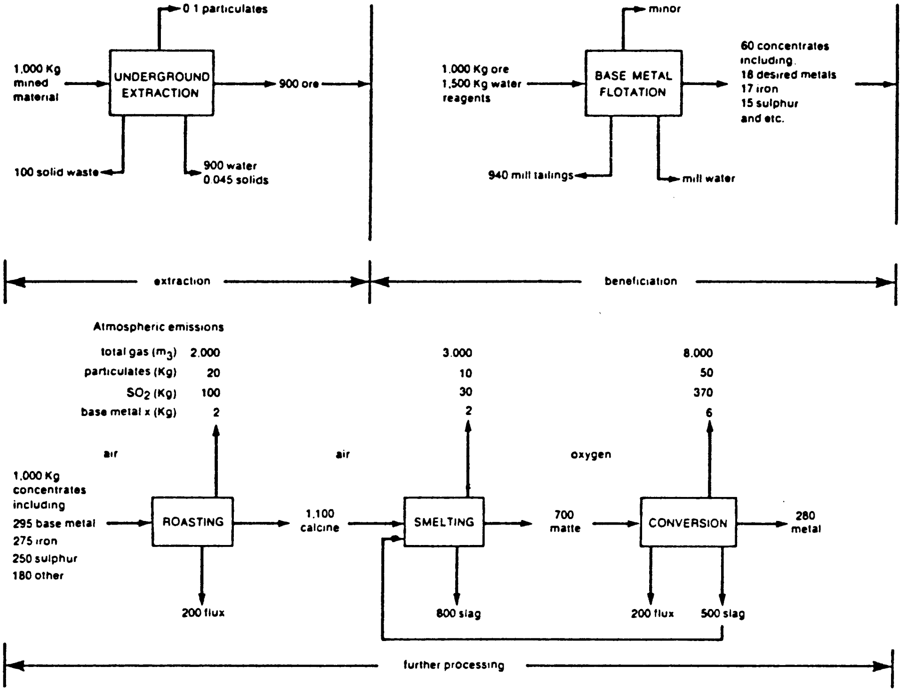

Fig. 2. Composite generalized materials flow diagram for the extractive, benefication, and further processing phases of a typical base-metal mining operation. (After Marshall, 1982)

2 Impacts of Terrestrial and Aquatic Disposal of Mine Tailings

2.1 Terrestrial Disposal of Mine Tailings

Acid mine drainage (AMD) is generated by chemical reactions and bacterial oxidation processes. Sulphide ores contain large quantities of pyrite which is discarded in the tailings and produces sulphuric acid (AMD) when exposed to water and oxygen. The ferrous iron produced is then oxidized to ferric ions which become the dominant oxidizing agent of the exposed sulphide minerals. The process thus becomes auto-catalytic in the production of AMD. In addition, the reduced sulphur and iron compounds in the deposits are excellent ecological niches for *Thiobacillus ferrooxidans* which oxidizes ferrous iron, thiosulphate, sulphur and metallic sulphides to obtain energy for growth while using oxygen as the final electron acceptor, using CO_2 as its sole source of carbon and requiring an environment with an acidic pH. Other species with related or similar needs and abilities are *Ferrobacillus ferrooxidans, Ferrobacillus sulphooxidans*, and *T. thiooxidans*.

The eventual pH of surface or groundwaters receiving AMD depends on their original alkalinity and the rate of AMD production. Natural alkalinity may keep the receiving waters neutral for some time, but there may be a rapid shift to acidity, usually difficult to predict, depending on hydrological and geochemical factors. For example, the production of acidity and alkalinity differs greatly in relation to the frequency of drying out and flushing of land disposed mine waste. Laboratory tests to predict acid versus alkaline production and thus the potential AMD are used extensively and include the use of kinetic titration tests, *Thiobacillus ferrooxidans* in inoculated, incubated shake flasks; soxhlet extraction procedures; and lysimeters; amongst others.

2.2 Subaqueous Disposal of Mine Tailings

Subaqueous disposal of mine tailings is employed to avoid terrestrial AMD. While land disposal of mine wastes is ideal for *T. ferrooxidans*, under water disposal in lakes or coastal marine sites makes colonization by the bacteria difficult if not impossible. There is less oxygen available in aquatic systems and it is unlikely that pH could drop low enough to accelerate *T. ferrooxidans* activity. In addition, chlorine in marine systems is toxic to the bacteria. However, there are negative effects of subaqueous tailings disposal related to turbidity, sedimentation, toxicity and contamination. The tailings particles may effect filter feeding or respiratory structures, cause gill damage in fishes, or increased oxygen consumption rates, along with other problems. Rapid tailings deposition causes burial of organisms. At lesser rates of sedimentation, benthic organism food supply to is reduced, habitats are affected, or upstream movement of biota can be physically blocked. In addition, many components of mining wastes are lethally or chronically toxic to fish or cause sub-lethal effects. Phytoplankton, periphyton, macrophytes, zooplankton, zoobenthos, and fish to varying degrees are effected by one or another of metals such as As, Cd, Cr, Cu, Fe, Pb, Hg, Zn, Ni, Sn, mixtures thereof and cyanide as well as other materials in mine effluents. There area a vast number of references relate to these toxicity aspects of metals to freshwater biota (Rescan, 1989) and there is no value in listing them here. The basic point is that mine effluents and emissions have been documented to cause both lethal and chronic effects on a range of aquatic biota, depending on factors such as the composition of the effluents and the receiving waters. There is also a vast literature on the uptake by biota, bioaccumulation and biomagnification of many mining related metals including Zn, Pb, Ni, Cd, Cu and Hg.

Assessment of sediment (tailings) associated metal impact on biota requires a knowledge of the bioavailable fraction of the metals in sediments because the total metal content is not the biologically important concentration. Chemical fractionation of sediments can be used to assess this bioavailable fraction. A series of reagents can be used to sequentially extract correctly collected and stored sediments and determine this bioavailable fraction. The exchangeable or loosely-held fraction can be removed by extractants such as ammonium acetate, ammonium chloride and magnesium chloride. The carbonate faction can then be removed by extractants such as acetic acid and sodium acetate. The easily reducible fraction which is mainly associated with new Mn oxides, can then be removed by extractants such as sodium acetate/hydroxylamine hydrochloride and nitric acid/hydroxylamine hydrochloride. The moderately reducible fraction, which is mainly associated with aged Mn and Fe oxides, can then be removed by extractants such as acetic acid/hydroxylamine and ammonium oxalate/oxalic acid. The organic and/or sulphide fraction can then be removed by extractants such as hydrogen peroxide/nitric acid and hydrogen peroxide/ammonium acetate. Finally, the residual fraction, mainly associated with the matrix, or lattice structures of sedimentary minerals, can be dissolved by reagents such as perchloric acid/hydrofluoric acid and nitric acid/hydrofluoric acid. These sequential metal extraction procedures for sediments can also be used to predict the remobilization of metals under changing environmental conditions.

In Canada, the Federal Metal Mining Liquid Effluent Regulations prohibit subaqueous disposal of mine tailings but exemptions are sought and given and subaqueous disposal of mine tailings occurs into both freshwater and marine environments. Whether underwater disposal of mine wastes is less of an environmental insult than terrestrial disposal is a subject of ongoing study and some debate. To resolve the questions raised will require extensive research at sites of former tailings disposal in lakes and at sites where this is still going on. The factors which play a major role on the fate of heavy metals in bottom sediments (and deposited tailings) are oxidation state; benthic organism type and density; sedimentation rate; type of sediments, e.g., oxyhydroxide phases; and organic matter accumulated. Lake sediments are chemically zoned with depth, and oxygen is depleted at some depth below the sediment-water interface. In mesotrophic and eutrophic lakes, the organic carbon content usually results in anoxic conditions at sediment depths of a few millimetres to 10 cm. The input of inorganic mine tailings to lakes dilutes the organic content of the natural sediments and buries this more rapidly. With rapid tailings impacts, the smaller quantities of oxygen buried are more rapidly, used up, and anoxic conditions established at shallow sediment depths. However, bioturbation and physical processes in both marine and lacustrine situations compound the basic geochemistry by mixing oxygen to depth in the sediments, causing oxidation of sulphides and release of metals to the pore waters. Seasonal limnological processes, such as turnover, stratification and productivity also affect oxygen content of bottom waters and thus affect the oxic zone thickness and resulting transfers of bioavailable metals.

The amorphous iron and manganese oxides and hydroxides which can make up to 10% by weight of lake sediments, are important scavengers of heavy metals because of their high specific surface area; high negative surface changes; and high cation adsorption capacity over the pH range of most natural waters. Dissolved metals may diffuse upwards in sediments with thick oxic layers but diffuse into sediments where anoxic sediments are at shallow depths. Thus elevated bottom sediment surface concentrations of Cu, Ni, Zn, Cd, Pb and Mo may not always reflect anthropogenic inputs if the pH of the overlying water is low. At polluted sites, metals diffuse into the near surface anoxic sediments to be precipitated as sulphides whereas at unpolluted, oxygen-rich sites, metals in sediment porewaters may be released to the overlying water. Under anoxic conditions in sediments, most heavy metals should be present as precipitated, insoluble sulphides, e.g., ZnS, HgS and FeS (thought be a significant sink of heavy metals in anoxic sediments). These considerations of the general behaviour of metals in lakes should be used to assess the efficacy of sub-aqueous disposal of metal-rich tailings. When tailings are to be dispersed into water systems, be they lakes, rivers, reservoirs, or coastal areas, then a great deal of basic information on the site is needed. Many of the effects will depend on the depth and size of the water body, flow, flushing rates, natural geochemical conditions, currents and many other site specific variables. In lakes, stratification, mixing patterns and biotic transfer mechanisms will be very important and for large rivers, mixing patterns and side effects.

3 General Principles of Mitigation and Remediation

Remedial measures to reduce metal releases to terrestrial environments should first be applied near the source of their release. Solid wastes on land can be limed or encapsulated by use of impermeable base liners and surface covers to reduce AMD and related metal movements and resulting biological availability of the metals. Achieving a neutral or slightly alkaline pH by application of lime, limestone, alkaline fly ash and similar materials favours adsorption or precipitation of metals.

Limestone liners for landfills may be an economical chemical barrier to precipitate solubilized metals in AMD. If the initial interstitial water in wastes is reduced, lower concentrations of metals are found in the leachate and an impermeable lining or an elaborate leachate treatment system may not be needed at the disposal sites.

In general, control of AMD is more difficult than prevention but mitigation of AMD can include: biologically enhancing AMD to remove sulphides, collecting the leachate and removing the metals; covering mine wastes with lime or limestone; diverting water inflow to the waste or from the waste; capping the wastes with clay or other material or flooding the surface to reduce oxygen ingress; using bactericides to kill bacteria that cause AMD; and applying phosphate to the surface to break the ferrous/ferric/pyrite cycle. Such procedures can be costly and only have varying degrees of success. For example, diverting water, then applying lime and sewage sludge followed by re-vegetation resulted in only a modest improvement in AMD from gold mine wastes after 10 years (Hinkle, 1984).

In aquatic systems, contaminated sediments can be isolated from overlying waters by physical barriers such as polymer films, or covered by sand and gravel. Waters can be limed. The best way to prevent problems with metal-rich wastes in aquatic systems is by isolating them in a permanently reducing environment. In general, the marine sulphidic environment is more favourable for metal-rich waste disposal than freshwater lakes due to the high stability of metal sulphides. Deposits above the marine floor can be capped or the waste placed in subaqueous depressions which are then covered by clean sediments.

Table 2. Authorized levels of substances in mine effluents

Parameter	Authorized Concentrations[a]			
	Maximum acceptable arithmetic monthly		Maximum acceptable in grab sample	
Arsenic	0.5	mg/L	1.0	mg/L
Copper	0.3	mg/L	0.6	mg/L
Lead	0.2	mg/L	0.4	mg/L
Nickel	0.5	mg/L	1.0	mg/L
Zinc	0.5	mg/L	1.0	mg/L
Total Suspended Solids	25.0	mg/L	50.0	mg/L
Radium-226	0.37	Bq/L	1.11	Bq/L
pH	6.0	or greater	5.0	or greater

[a] All concentrations are total values except for Ra-226 which is at the dissolved level after filtration of the sample through a 3-micrometre filter. The levels shown apply to undiluted effluent at the point of discharge to the environment.

In Canada, the Federal Metal Mining Liquid Effluent Regulations and Guidelines give the authorized acceptable levels for substances in mine effluents (Table 2). The federal regulations in Canada are for base metal, iron and uranium mines and only apply to new, re-opened or expanded mines after 1977. Provincial requirements apply where they are more stringent and site specific regulations may also be applied. However, regulations need to be enforced, not simply developed and ignored as is the case in many developing and even in some cases developed countries.

3.1 Mine Wastewater

Processing of ore grade metals introduces a variety of chemicals to the mine waste streams (Table 3) including nitrogen compounds from blasting operations, colloidal suspended solids, many other trace metals of non-ore grade (usually Al, As, Cd, Cu, Fe, Pb, Mn, Ni, Zn, Rd), and thiosalts ($S_2O_3^{2-}$; $S_3O_6^{2-}$, $S_4O_6^{2-}$) which on further oxidation cause acidification of rivers and lakes. Wastewater treatment most commonly uses neutralization and concurrent precipitation of the metals as hydroxides under alkaline conditions. The most frequently used reagent is hydrated lime ($Ca(OH)_2$), often produced on site by slaking of quicklime (CaO). Multistage precipitation is used for different metals and specific to the metal content of the ores. Solid/liquid separation occurs between treatment stages and difficulties can arise in separation of the amorphous sludges. Cyanide needs to be removed from the effluent of precious metal mines (Au and Ag). NaCN is used to complex the Au which is then replaced by Zn or extracted directly by sorbtion onto carbon granules. Both processes release highly alkaline effluents rich in CN ions

Table 3. Typical ore processing reagents

Reagents	Comments
Acids (H_2SO_4, HCl, HNO_3)	-
Alkalis (CaO, $Ca(OH)_2$, $CaCO_3$, Na_2CO_3, NaOH, NH_4OH, NH_3)	-
Frothers and Collectors	Surface active organics
Modifiers	Surface active organics and various inorganics such as NaCN, Na_2SO_3, $CuSO_4$, $ZnSO_4$, Na_2S, $AlCl_3$, $Pb(NO_3)_2$, silicates and chromates
Sodium Cyanide	Used for precious-metal cyanidation and as a depressant in flotation processes
Flocculants, Coagulants	Aluminum and iron salts, and organic polymers

and various cyanide containing complexes. Gold mines use many ways of removing CN including natural degradation; alkaline chlorination; various SO_2 processes; hydrogen peroxide oxidation; anion exchange; and biodegradation. Radium in mine wastewaters can be removed by precipitation with barium sulphates. Thiosalts are removed in effluents as sulphur, sulphides or as H_2S gas.

3.2 Contaminated Groundwater

Effects of mining on groundwater can be rapid or long-term (Loxham, 1988). Predicting effects requires a detailed knowledge of groundwater characteristics, flows and volumes as well as of the materials in the tailings and those onto or into which the tailings are introduced. Control options include preventing the infiltration of contaminants; stabilizing the contaminants chemically; or removal and treatment of the contaminated groundwater. One option is to attempt to cut off the movement of contaminants but most such options are not long-term. Cut-off walls can isolate a leachate so that it can be pumped out and treated but the walls must extend to an impermeable layer beneath the aquifer. If such layers are deep, cost can prove prohibitive, as they can be to seal the bottom. When wells are used to remove a leachate, a difficult choice has to be made on the number of wells and, based on the expected contaminant concentration, the cost of treatment of the water volume expected to be removed. However, given adequately constructed cut-off walls, and shallow depths to an impermeable barrier as well as removal and treatment of leachates, a long-term, relatively safe scenario can be developed.

3.3 Contaminated Sediments

Techniques to treat metal contaminated sediments should be based on the short and long-term leachability of the metals. Dredging, compaction (dewatering and hydrocyclonage), and disposal are the normal procedures but in some cases additional decontamination steps can be used such as leaching, ion exchange, extraction or flotation techniques. Hydrocyclonage has a limit of separation of some 10 mm but is large capacity and costs are low. Flotation skims off air bubbles attached to particles because of their hydrophilicity. High gradient magnetic separation techniques can be used on particles larger than 1 mm and separate ferro and paramagnetic compounds; metals coprecipitated with ferric hydroxide; and non-magnetic particles bonded to a magnetic floc. These

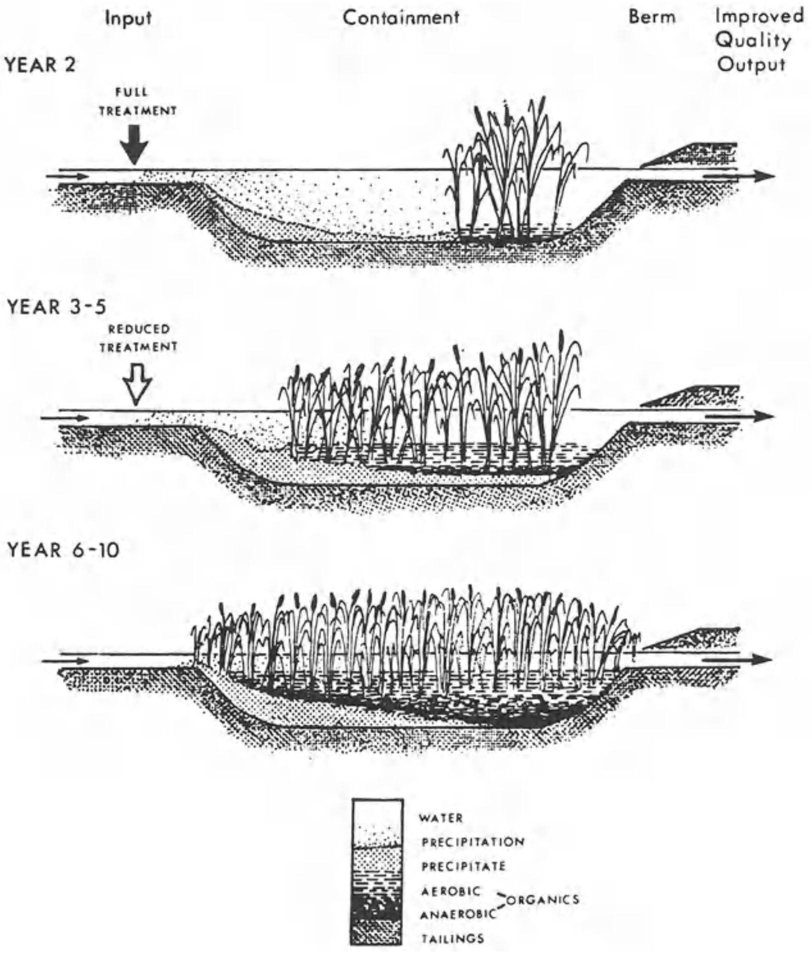

Fig. 3. Schematic development of wetlands on waste sites. The first 2 years might require some neutralization treatment to assist in the establishment of suitable indigenous species. As establishment of the wetland vegetation increases throughout years 3 to 5, the treatment can be reduced and is gradually withdrawn within years 6 to 10 when the system is self-sustaining. The anticipated sediment development is depicted in the enlarged insert providing an anaerobic, aerobic and finally precipitate layer underneath the treated water. (Kalin and van Everdingen, 1988)

techniques can be used on a large scale. Small scale techniques include removal by leaching, e.g., Hg with sodium hypochlorite when the Hg is present as a sulphide or an organic compound, e.g., in anoxic lake bottom sediments. Heavy metals can be dissolved by acid leaching at pHs of 0.5 to 1.0 with separation of the acid solution and precipitation of the metals with lime, or ion exchange can be used but requires long contact times. Sediments can be pretreated by formation of flocs by use of agents such as $FeCl_3$, $Ca(OH)_2$ and polyelectrolytes and then dewatering by sedimentation or filtering, or dewatered directly by electro-osmosis.

3.4 Passive Biological Techniques

Self maintained, passive biological systems can be employed to treat mine wastewater and waste leachates. These methods are attractive because of the low cost. Systems proposed include use of mosses which can reduce infiltration of precipitation into terrestrial mine wastes or remove metals such as Cu and Ni from leachates or wastewater, if flow rates are low. Wetlands (Fig. 3) are now being studied as to their value in reducing acidity and heavy metals but their use for biofiltration is not yet widespread. Cattails can be introduced to alkaline mine slimes by hand transplanting or by mechanical means. Seeding cattails show varying degrees of success and burial of the rhizomes under compacted material deeper than 10 cm inhibits expansion. Bacteria or fungi can concentrate and remove metals from mining effluents. Some macroscopic attached algae (*Charophytes*) have been tested in alkaline waters in polishing ponds at uranium and nickel mines and concentrated several metals and increased pH to neutral conditions.

4 Specific Examples of Mitigation, Remediation and Lack Thereof

Mitigation and remediation are not absolute concepts. There is now an extensive literature base describing metal contamination at sites around the world. At most, sites, some basic attempts at mitigation are employed. There is a complete spectrum from virtually zero mitigation to extremely well planned long-term mitigation-remediation programs. The following examples show this wide range of situations from local to regional; from well managed where regulations are strictly enforced, to where less controlled situations can lead to major impacts; and various remediation procedures as well as situations where clean-up is now virtually impossible.

Fig. 4. Location of the main sites mentioned in the text

4.1 Flood Plain Contamination and Remediation

Flood plain contamination and long-term, long range effects are seen in the Clark Fork River system U.S.A. (Fig. 4). The headwaters area of the river include the Butte District Cu mining and smelting complex which opened in the late 1800s. Smelting ceased in 1980 after more than 1 billion tonnes of ore had been removed and primary wastes covered 35 km^2 with associated heavy metal contamination from smelting covering hundreds of km^2. Although metal concentrations in river bed sediments declined rapidly downstream from the mining area, the concentrations for some metals, some 380 km from the primary sources were still an order of magnitude higher than in (non-contaminated) tributaries (Luoma, 1989). This long range effect was partly related to the storage of metals in the flood plain soils which became a secondary source of metal input as new river channels were cut. Flood plain soils often had higher Cu and Zn concentrations than the adjacent river bed sediments. Concentrations trends of metals in various biota for up to at least 380 km downstream closely paralleled the river bed sediment concentrations. Such extensive contamination of the flood plain for hundreds of km, virtually precluded any remediation for technical reasons and cost.

A similar situation but one feasible to partial remediation occurred in the Jinzu River mining area of Japan (Fig. 4). Ore deposits of Zn, Pb, Au, Ag and Cd were found some 1,200 years ago and were rapidly developed from 1890 on. Over 4,000 tons of ore were mined daily at peak productivity and smelting occurred at the site. Downstream river paddy soils became contaminated by Cd and the area experienced Itai-Itai disease from 1971 to 1976. The remediation technique used was to cover the most polluted soils with 20 to 30 cm of unpolluted soil at a cost of some $80,000 U.S./ha (1978). Rice grown in the clean soil cover had Cd concentrations below that in rice from unpolluted paddy fields. Measures were also implemented to reduce effluents from the mines and smelters so that the problem did not repeat itself. Regional soil pollution from smelter emissions was also well documented but was so widespread that remedial action for such soils was not considered.

4.2 Subaqueous Disposal of Mine Tailings and Remediation

Benson Lake, in British Columbia (Fig. 4) received tailings from a copper mine between 1962 and 1973. Some 3.6 million tons of ore were mined and the tailings discharged to the lake under permit. Tailings were introduced to the lake at a depth of about 43 m. The central and deepest part of the lake was just over 50 m and the lake bottom became covered with tailings. By 1967, no benthic organisms were found except near the lake outflow. The main problem was seen as turbidity from colloids which did not settle out and which appeared at the surface once lake stratification broke down. The mitigation measure which proved generally unsuccessful, was the extension of the tailings pipe to some 48 m. Buttle Lake, also in British Columbia (Fig. 4) also received tailings from several mines with recoverable quantities of Cu, Zn, Pb, Au, Ag and Cd. Buttle Lake is 30 km long by 1.5 km wide with a maximum depth 87 m, in an area of high relief. Some 5.5 million tons of mill tailings were deposited in the lake between 1967 and 1984. Prior to discharge into the lake, sand size particles were removed from the tailings and back filled into the mine workings. The discharge outfall was 30 m below the thermocline and a flocculent was used to promote settling to the bottom some 30 to 50 m deeper than the discharge point. The pH in the milling circuit was kept high by additions of lime. The permit levels for the dissolved metals were 0.3 ppm Cu, 0.1 ppm Pb, 5.0 ppm Zn and levels achieved with a chlorination plant were 0.09 ppm Cu, 0.08 ppm Pb and 0.43 ppm Zn. Analyses of sediment cores and their porewaters from natural sediment and tailings areas showed that remobilization of metals was minimal and concentrations for some metals were lower than the overlying bottom waters. Oxidation of the tailings did not appear to be taking place due to the rate of tailings discharge (rapid burial). There was no release of Zn and Cu and these metals were diffusing into the natural sediments and being precipitated as sulphides because anoxic conditions occurred at shallow sediment depths. When tailings discharge ended in 1984, natural sedimentation of some 0.2 cm per year bury the tailings to a depth of some 3 cm by 1999. This should prove sufficient to have anoxic conditions at the 3 cm sediment depth or less and thus seal the tailings. When high Zn and Cu levels were found in the lake water and it was proposed that these came from the deposited tailings, studies discovered the power to be a creek draining a waste rock site subject to AMD. Once this source of AMD was eliminated by leachate collection and treatment after 1983, the concentrations of Zn and Cu in the lake water began to drop and by 1990, they were near pre-mining concentrations of <5 ppb Zn and <1 ppb Cu. Today, the mine continues to operate, but the tailings are now stored in ponds near the mine, the dissolved metals in the discharge water removed by liming, and the water then recycled to the mill. It is extremely rare that an ore body is conveniently discovered near a meromictic lake, but this was the case for the Polaris Pb-Zn mine in the Canadian High Arctic. Tailings are being deposited at depth into meromictic Garrow Lake (Fig. 4) and no deleterious impact on the surface waters or outflow have been recorded.

Turning to the marine-coastal situation, the Pb-Zn Black Angel Mine is located on a fjord system on the west coast of Greenland (Fig. 4). Since 1973 approximately 500,000 tons/year of metal rich tailings has been discharged in a salt water suspension into Agfardlikavsa Fjord at a depth of 30 to 42 m. The tailings are 50% marble/dolomite and 50% pyrite. The effects are Zn, Cd and Pb contamination of the water and suspended particulate matter in the fjords (Loring & Asmond, 1989). Dissolved Zn and Pb reached levels of 1 ppm at the bottom of the fjord and metal enrichment in biota occurred up to 50 km from the effluent input. Seaweeds and mussels in the fjord were contaminated by Zn. Seasonal stratification played an important role in the transfer of particulate and dissolved metals and during the winter mixed conditions, metal-rich particulates were transported over the fjord sill into an outer fjord where they were deposited in decreasing amounts seaward. Attempts to remediate such underwater tailings at marine disposal sites are virtually non-existent (Ellis & Taylor, 1988). Whereas coal mine open pits in areas of high land values, such as the brown coal mines pits near Cologne, Germany (Fig. 4), are often restored to a semi-original state, the same does not apply to subaqueous lacustrine or coastal disposal which is out of sight. However, reclamation procedures could involve construction of artificial habitats with concrete, car bodies, or tires, or other waste materials that are eventually colonized to create new ecosystems. One benefit is that these habitats can even be constructed from other wastes, for example blocks made from the waste from coal-fired power plants even processed mine tailings. Attempts at nearshore revegetation, for example, by shoreline algae, have varying degrees of success. Colonization of tailings by marine benthos has only been studied rarely. The ability of benthos to colonize subaqueous tailings deposits depends on the tailings toxicity and rate of burial by unpolluted sediments. At the Island Copper Mine in B.C. (Fig. 4), some 106 million tons of tailings were deposited in two fjords between 1960 and 1981. Some 50 km^2 are covered by tailings averaging 15 m in depth and a maximum of 35 m. Some polychaete worms and some clams have been able to grow on the tailings and form a new benthic population or rather to survive. Other organisms that appeared able to colonize on or near the tailings are oysters and crabs. Closer to natural conditions can be reached after about two years once the tailings have stabilized and turbidity is reduced. Fishing often occurs at sites of marine disposal of mine tailings including near Island Copper. Natural recovery occurs but could perhaps be accelerated by fertilization or capping of the submerged tailings.

4.3 Natural Controls and Modern Mitigation

At new mines, even at remote sites in developing countries, impact assessments and implementation of mitigation measures and ongoing remediation can be made part of the cost of metal mining. The OK Tedi Mine in Papua New Guinea (Fig. 4) has become one of the most studied sites of tailings discharge to a large tropical river system (Salomons & Eagle, 1990). By 1989, some 150,000 tons of Cu-rich ore was milled each day. Tailings and other waste materials are discharged to the OK Tedi, a tributary of the Fly River which is the largest river in Papua New Guinea and which discharges to the Coral Sea. Questions about potential pollution of the Coral Sea and impacts on the Great Barrier Reef resulted in analyses of the fate of the metals in the tailings. Most of the Cu-rich tailings have particle sizes <100 m and are thus transported down the entire river system, a distance of over 1,000 km before being largely deposited in quiet coastal zones. The natural carbonate-bicarbonate geochemistry of the river system buffers acidity. Because of the high dissolved organic carbon geochemistry of the river, Cu remains largely in the suspended sediments. Some dissolution of Cu occurs in the coastal marine environment. Since 1981, a progressively more complex monitoring program has been used to assess the impact of the tailings discharge. Hydrological, geochemical and biological measurements included cation exchange capacity (CEC) of the suspended sediments and the water complexing capacity for Cu. Copper-enriched tailings are diluted downstream by natural sediments with a higher CEC capacity. Modelling of dissolved Cu has taken into account alkalinity, pH and dissolved organic carbon and predictions closely match measured concentrations.

The Navan deposit in Ireland (Fig. 4) is 80 million tons of Zn-Pb located 50 km north-west of Dublin. Modern technology and mine practices are employed (Aldwell, 1990). All aspects of soil and water contamination by Zn, Pb and Cd are considered. Various European Community environmental directives came into effect during the opening of the mine in 1977. Dust with elevated Pb concentrations was considered a worse problem for local human health than waterborne contamination of the nearby rivers. The dust problem was found to be mainly related to drying out of the tailings in spring due to frost and strong winds, a situation probably unique to this site. The water quality standards to be met after any effluents were released were those recommended for salmonoid rivers. There was a specified sampling regime for effluents and the river, including fish analyses for metals and population surveys. The main sources of water pollution are as usual, mine drainage; surface drainage from the site; mineral processing effluent; and tailings discharge. Water is stored and recycled and the final discharge to the River Boyne is by a 31 m long distributor on the bed of the river and is altered depending on river flow. A three-stage tailings pond is some 5 km from the river. The tailings are kept saturated, drains intersect seepage, and areas of the dam floor that need it have a clay liner. Observation wells allow metal analysis of seepage. Rehabilitation of the tailings involves revegetation with Zn and Pb tolerant grasses and is expected to produce a permanent vegetation cover without the need for a surface soil layer. Everything at this site is being done to reduce effects and to monitor all possible contamination in a manner that could result in immediate action to reverse it. Any complaints by the local population of the surrounding farming community are dealt with immediately.

4.4 Post Mining Leachate Remediation

The Rum Jungle mining area is in the tropical Northern Territory of Australia (Fig. 4). Uranium and copper were extracted from a number of ore bodies. After the mine closed in 1971, aquatic flora and fauna in the river below the mine site were adversely affected by trace metals. Waste rock dumps and the copper heap leach were found to be the dominant sources of Cu. Calculations of annual sulphate release and sulphur present showed that the waste dumps would continue to be a source of AMD and heavy metal pollution for hundreds of years. The rehabilitation strategy developed (Harries and Ritchie, 1988) consisted of covering the dumps with a seal to reduce water percolation; collecting the tailings and covering them with enough material to reduce radon release (because this was partly a uranium mine); removing the copper heap leach pile and disposing of the material above the water table in a location with minimal water ingress; treating the water in the open cuts to raise the pH and reducing the metal concentrations to a recreational standard where limited nose and ear contact with the pit water would be permissible; and cleaning up and revegetating the other areas.

4.5 Mine Tailings Revegetation

A major effort at revegetation has been underway for many years at Sudbury where the Inco Limited mining, milling, smelting complex is one of the world's largest (Fig. 4). Tailings were deposited in the valleys but as the tailings piles grew, dust control became a problem. A series of bituminous sprays, limestone chips, water and chemical sprays and snow fences were used to mitigate the problem. In the 1950's, a revegetation research program began using various local grasses and other plants (Peters, 1988). The tailings were pre-treated by mixing in limestone and fertilizer. Other additions tried with varying success were wastes from cement, paper, chemical and sewage plants. The mixing of wastes to achieve a benefit is a valuable concept but costs can be prohibitive due to transportation. Irrigation was used when needed.

Over the 30-year program, several key guidelines to revegetation were developed. These included adding limestone to raise pH to 4.5 to 5.0 before seeding; planting be at a time of adequate water supply; introducing seedlings closest to the prevailing winds to prevent their burial; creating rapid colonization; using combinations of plants to provide shade for grasses; using mulch to provide shade and to reduce moisture loss; reducing moisture loss on slopes by clay covers using local species of plants; applying nitrogenous fertilizer frequently; and using nitrogen fixing plants if possible.

By the mid-1960's, local birch trees and some aspen and willow had invaded the grassed areas. In the early 1970's, several types of trees were subsequently tested. Jack pine proved the most successful. Cattails, bulrushes and mosses became established around ponds. Insects, smaller mammals and birds followed. Uptake of heavy metals by these animals was investigated and not considered a problem. Finally, Canada geese were introduced in 1985.

Revegetation mitigates effluent water quality because plants intercept precipitation, transpire it, and form an oxygen consuming barrier, reducing AMD and thus heavy metals in leachates. These general procedures and principles applied at the Sudbury area could be the basis for similar revegetation attempts elsewhere. Specific additions of limestone, fertilizer, or rock phosphate are required depending on local conditions, as well as the composition of the species mix. However, at any site, much is on a test, trial and error basis and must usually be long-term to guarantee a satisfactory outcome.

5 Regional Remediation

Regional pollution related to mining activities, other than by aquatic transport, is usually the result of airborne dispersion. Locally, dust from tailings or other waste sites can be a problem and this was mentioned above. Regional airborne dispersion occurs from the stacks of smelters and roasters. At such sites there are tailings or waste piles and local soil pollution and groundwater contamination. These local problems can be remediated by the procedures described earlier. The regional scale is outside the plant fence, perhaps several kilometres out from the plant. The outward extent of contamination can be tens to hundreds to even thousands of kilometres if the smelter has a super-stack. Other than cutting off all emissions, there is little that can be done to reduce long-range, low-level impacts of contaminants in smelter emissions. However, at the medium regional scale of tens of kilometres around the plant, some forms of pollution can be remediated.

The first step in regional remediation is to assess the extent of the contamination. Pollution from a smelter could involve acidification and heavy metal contamination of waterways, particularly lakes, and soils, and impacts on crops and vegetation, for example, forests. Pollutant transport models can be used to define the most polluted areas or actual analyses of soils, vegetation and water bodies to determine the extent of the most contaminated sites. Once the most severely acidified or heavy metal contaminated water bodies or soils are identified, remedial options similar to those described above for local problems can be considered. The impacts of the pollutants vary, of course, dependent on the natural geochemistry of the soils and water bodies, namely their buffering or metal sorbtion capacity. Clean-up will be a priority if urban areas or villages are within the highly polluted zones, especially if human health is directly affected. It may be necessary to replace or treat garden soils used to grow food. If, on the other hand, the major impact is in a wilderness area, the priority may be lower for expensive *in situ* remediation.

The options that can be used regionally, range from doing nothing to closing the smelter. Intermediate remedial options are to define the most polluted sites in a socio-economic content and to clean these up. Remediation could include treatment of contaminated drinking water reservoirs to removing metals; setting fish consumption advisories for water bodies; liming of lakes or soils; covering of contaminated soils with new top soil; immobilizing metals in the soil by adding clay or manure to increase sorbtion capacity; *in situ* treatment of soils to remove heavy metals; removal and disposal of contaminated soils or sediment; deep ploughing of soils to lower surface metal contamination; leaching soils with irrigation techniques and perhaps collecting and treating the run-off.

The above options should only be pursued if accompanied by application of the best available or practical technology at the emission source. Application of best available technology can be a costly hard option. Soft options are to better educate the smelter operators; to inform the operators of the regional pollution consequences, even to their own families, of mistakes or carelessness; better discipline at the plant; good housekeeping; preventative maintenance; and quality management. All of these low cost, soft options can result in emission and regional pollution reductions. Some of the options to reduce emissions are cost effective. For example, SO_2 can be converted to sulphuric acid and sold and Cd can be removed and sold.

Various land use alternatives should be considered. For example, crops and animals for human consumption can be raised on the least contaminated zones; for some animal feed (not for human consumption) on the more contaminated sites; and the most contaminated sites used for crops to produce textiles or for non-agricultural uses. Some metal hyperaccumulator crops can be used on a regional scale to bioaccumulate metals from the soils and help to decontaminate them. If groundwater has been contaminated, then the remediation procedures described earlier can be considered. These involve hydraulic curtains; extraction and treatment; isolation of aquifers; and regulations on aquifer uses. If a considerable area is severely contaminated, a final drastic option is simply to abandon the area, move the people and let it return to nature as best it can.

6 Remediation to Pre-Mining Status

In Canada, the last twenty years has seen a great increase in the number of lake and stream sediments systematically sampled and analyzed for heavy metals for ore deposit exploration purposes. By using GIS systems to link these individual surveys, maps of extremely large areas can be produced. The sampling procedures for the Canadian Shield were such that the concentrations reflected background conditions and excluded anthropogenic influences such as mining or smelting.

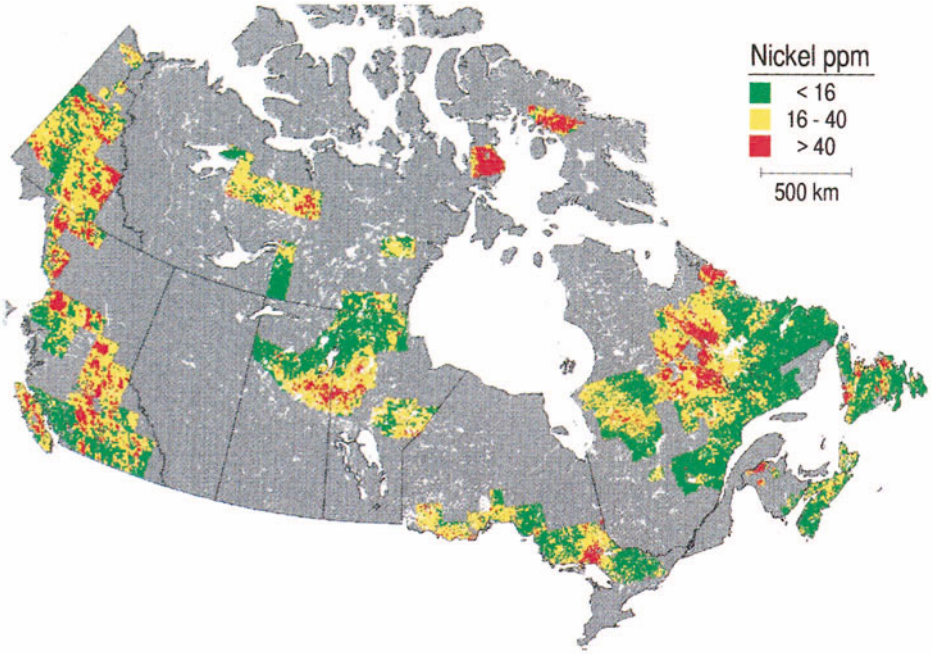

Fig. 5. Nickel distribution in aquatic sediments in Canada

The Ni anomaly associated with the Sudbury area is clearly seen north of Lake Huron (Fig. 5). Similar naturally elevated concentrations are found for other heavy metals in other parts of Canada.

When ore deposits are discovered, they often have surface outcrops which are already oxidized and leached and already have heavy metal rich, acidic runoff resulting in heavy metal anomalies in downstream rivers and lakes. Such weathering may have gone on for thousands (glaciated) to millions (tropics) of years and the vegetation and biota at these sites adapted to the conditions. In all cases, local and regional, consideration of the natural heavy metal concentrations in soils, sediment, water and effects on biota at mineralized sites should be considered as a baseline for remediation. Most metal-rich waste sites are hostile environments for most plants because of toxic metal concentrations; low plant nutrient levels; physical structure in the wastes; and variable oxygen access to the wastes. However, vegetation and biota that are metal tolerant often occurs at mine sites in spite of existing geochemical anomalies related to the ore deposits and such plants may be useful in mine waste remediation.

7 Conclusions

Terrestrial and aquatic ecosystem contamination by acid mine drainage and heavy metals is a global phenomenon. In some developing countries, there are often no environmental restrictions on mining activities or if they exist, are seldom enforced. The same now appears to have applied under the former regimes in eastern Europe and the Soviet Union and may still apply to remaining communist states. In the industrialized nations, the situation, although normally much better, is sometimes far from satisfactory. Many mines have operated for years, even centuries, with few restrictions. Regulations have been applied only recently and often only to new mines.

The basic problem with mining is the enormous volume of the wastes. Disposal on land has been the historic procedure but it is clear that in nearly all cases this results in acid mine drainage and heavy metal dispersal and contamination. Disposal of tailings or effluents into waterways injects the contamination directly. However, under specific conditions subaqueous disposal greatly reduces AMD but alternatively toxic metals are released or recycled or other impacts occur on biota. Given the ongoing development of legislation on mine wastes and disposal practices, it is likely that subaqueous disposal may not always be sanctioned, even when it avoids some terrestrial contamination problems.

The main steps in remediation of mining site pollution are to: (1) assess the extent of the local site pollution and quantify the sources of AMD, heavy metals and other pollutants; (2) control the major sources (waste dumps, tailings, air pollution, etc.). [Control effluents/emissions, reduce leachates, bury, lime, remove treat (leach, extract etc.)]; (3) assess the regional pollution status and identify hot spots. [Remediate hot spots - dredge rivers, lime lakes, treat *in situ* (create anoxic conditions), flood, bury, cover, etc.]; (4) leave to natural recovery or accelerate. [Increase erosion/deposition of clean sediments in bays/rivers, divert alkaline streams (AMD), eutrophy, flush, create wetlands, etc.] The terrestrial techniques that can be employed include covering tailings to reduce infiltration (soils, vegetation, clays, tills, polymers or cement like materials); artificial creation of hard pans (limonite, goethite, iron hydroxide) to reduce infiltration; or blending in alkaline wastes. Polluted soils can be removed (treater, better disposed of) given new soil covers (buried) or limed. Aquatic sediments can be dredged (dewatered, leached, treated, disposed of in new, well designed waste sites), buried, or treated *in situ* (limed), or left to nature to restore. The aquatic techniques include subaqueous disposal in general or creation of water covers (anoxic conditions), or wetlands (to leach, filter contaminants and create anoxic conditions). High technology is not often the best long-term solution. Low technology, in particular the evolving biotechnology that would be especially practical in less wealthy nations, may be the best solution at many sites.

Remediation of mine sites is often beyond the financial ability of all but major corporations. Vast open pits, and polluted soils, river, floodplains, lakes and coastal zones such as fjords abound, even in the most advanced environmentally conscious nations. Given the number of mines, historical and present, and their associated waste rocked tailings piles and dumps and zones of aquatic ecosystem contamination, remediation or at least mitigation are a monumental task. Hopefully, new mines - and this is far from true - will employ state-of-the-art mitigation procedures such that additions to the global mining pollution inventory are kept to a minimum. The general requirement is that the best remediation is prevention or at least economically feasible mitigation while the mine is in operation. A mine site is usually already mineralized and often a geochemical anomaly for heavy metals and a source of AMD. The geochemical characteristics of the site should be assessed in detail. Exploration - trenching, drilling - and construction can enhance these long before mining itself begins. Environmental controls need to be in place even during these pre-production stages and eventual environmental impacts should be carefully assessed. Future environmental problems should be mitigated or avoided in the design of the mine and its operation. Environmental controls should be built in as construction proceeds and mining gets underway. Retrofitting or eventual clean-up is likely to be far more costly, never mind the corporate image generated by a lack of environmental awareness. During all stages, but especially from the start of construction, all environmental impacts should be monitored.

During mining operations, impacts should be continually assessed against predictions, for example the changed geochemical characteristics of lakes or rivers, and adaptation made as required - perhaps less routine measurements and more process research becomes the key requirement. Remediation should be ongoing so that as much as possible is done while the mine is producing and generating income. For old sites at operating mines, retroactive procedures such as revegetation, leachate treatment and others should be tried. Where human health concerns occur, for example due to food toxicity from mining pollution or wildlife and aquatic resources are impacted, the will is often there to remediate. In other cases, the situation may well be left to nature to heal as best it can.

Open metal mine pits present distinct problems. When pits result from coal mining, there may be little of the deposit left. The pits can be filled, flooded, and subject to various remedial measures, although these may well be costly and fraught with questionable predictions as to future groundwater impacts. In some cases, where coal mining has involved vast open pits, for example in the brown coal area west of Cologne in Germany, the plans for remediation are carefully implemented in great detail. As the coal is mined, waste from other pits is used to fill them, the landscape and soils are restored, and even the villages mined around or moved and rebuilt. This enormous remedial effort is only possible with decades of forward planning and because of agricultural land scarcity and economics. However, metal mine open pits are different, in that the viability of the mine depends on fluctuating world metal values and supply and demand. When a metal mine open pit is abandoned, this usually does not mean that the metals are all removed, only those economically feasible to do so at that time. Metal mine pits are opened, closed, opened and may well be needed by future generations when techniques to extract lower grade ore are used or developed or when the economics change. Remediation of such pits may only add or even prevent re-opening. There is a question then as to just what degree of remediation is desired for metal mine pits.

Innovative techniques for *in situ* treatment may need to be attempted. For example, once sources are controlled, a polluted water may be decontaminated by a series of well planned flushing stages, with construction of temporary dams to collect contaminated sediments, or the flushed sediments (to lakes, reservoirs) may then be subject to accelerated natural burial by increased upstream erosion or by *in situ* generated eutrophication. Such scenarios have not been attempted because of the faster than predicted rate of natural recovery and the fear of making the situation worse. The latter often also applies to dredging of hot spots. Nevertheless, if the situation is bad enough and the health consequences serious enough, costly engineering solutions may have to be considered.

If wise regulations, based on good ecosystem science, are set, monitored by adequate indicators of ecosystem health, and enforced, then pollution from future mines will be substantially mitigated. Meanwhile, and for the foreseeable future, the implementation of adequate mitigation globally remains an issue as well as the requirements for remediation of existing wastes. The need for innovative remediation techniques will escalate as well the need for the research to arrive at these techniques and predict their outcome. However, in many cases, remediation is beyond engineering and economical feasibility and assessment of impacts may be all that is possible while nature attempts to self-cleanse.

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