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## REVIEWS AND ANALYSES

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### Environmental Impacts of Metal Ore Mining and Processing: A Review

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#### ABSTRACT

The impact of mining and smelting of metal ores on environmental quality is described. Mines produce large amounts of waste because the ore is only a small fraction of the total volume of the mined material. In the metal industry, production of Cu, Pb, and Zn causes the greatest degradation of the environment. Copper mining produces extensive mine wastes and tailings and Cu smelting emits approximately 0.11 Mg of S per Mg of Cu produced in the USA. Zinc and Pb smelters release large quantities of Cd and Pb into the environment. Metal smelting and refining produce gaseous (CO<sub>2</sub>, SO<sub>2</sub>, NO<sub>x</sub>, etc.) and particulate matter emissions, sewage waters, and solid wastes. Soil contamination with trace metals is considered a serious problem related to smelting; however, mining and smelting are not main sources of global metal input into soils. Other sources like discarded manufactured products, coal ash, agriculture, and transportation take a lead. Smelters are the main sources of atmospheric emissions of As, Cu, Cd, Sb, and Zn on a global scale and they contribute substantially to the overall emissions of Cr, Pb, Se, and Ni. A quantitative evaluation of the environmental health effects of mining and smelting is difficult because of the complexity of factors involved and lack of consistent methodology. Nevertheless, the case studies described indicate that negative health effects could arise from Pb mining and smelting. Risk assessment revealed that food chain contamination by Cd from soils contaminated by smelting is very unlikely under the western diet.

**M**INING and processing of metal ores can be important causes of environmental degradation. Metal ores are extracted by mining, which involves removal of rock from the ground. In general, deposits within 100 m of the surface are extracted from open pit mines and those at greater depths come from underground mining. Open-pit mining produces larger amounts of overburden (the waste rock that overlies the ore) than underground mining does. For example, the Bingham copper mine in Utah, one of the largest open-pit mines in the world, removes >225 000 Mg of rock each day, only 20% of which is ore (Kesler, 1994). About 90% of the ore mined in the USA is extracted from open pits, but only 50% of the ore in the rest of the world is mined this way.

The environmental concern in mining areas is primarily related to mechanical damage of the landscape and acid mine drainage (AMD). Mines occupy about 3700 km<sup>2</sup> in the USA (about 0.26% of the USA land area)

compared with 100 000 km<sup>2</sup> for national parks and wildlife refuges, and >385 000 km<sup>2</sup> for the wilderness system (Barney, 1980). Cumulative use of land by mining throughout the world between 1976 and 2000 will be about 37 000 km<sup>2</sup> or about 0.2% of Earth's land surface (Barney, 1980). Copper mines alone will increase the use of land by about threefold, from 41 000 ha in 1976 to 113 000 ha in 2000. About 60% of disturbed areas is used for the disposal of overburden, which accounts for about 40% of the solid waste generated yearly in the USA (Barney, 1980).

Although mines are classified on the basis of their predominant product, they may produce large quantities of other elements as coproducts. For example, Zn mines in the USA produce 72% of all Zn; 100% of Cd, Ge, In, and Th; and 3.1, 4.1, and 6.1% of all Au, Ag, and Pb in the USA (Minerals Yearbook, 1992). As a result, metal ore processing usually leads to the multi-elemental contamination of the environment.

In general, economically important mineral ores occur as metallic sulfides, which causes SO<sub>2</sub>, a contributor to acid deposition, to be released into the atmosphere in smelting operations. Despite the steady reduction of SO<sub>2</sub> emissions since the 1970s and particulate matter since the 1950s, smelting is still a significant source of gaseous and dust contaminants (Fig. 1 and 2). Copper smelting is regarded as one of the major manmade sources of SO<sub>2</sub>. The Sudbury Ni and Cu smelters (Ontario, Canada) produced more SO<sub>2</sub> between 1969 and 1979 than all volcanoes throughout the earth's history (Dobrin and Potvin, 1992). Consequently, soil near the three Sudbury smelters had pH values from 3 to 4 and lake water pH from 4 to 5 (Dixit et al., 1992a). The water pH has decreased by 1 to 2 units in Sudbury area lakes since preindustrial times as estimated by diatom analysis (Dixit et al. 1992a,b).

Topsoils in the vicinity of smelters contain elevated levels of trace elements (Merrington and Alloway, 1994). The abundance and diversity of soil microorganisms are generally reduced by smelter emissions and acidic soil conditions. This reduction lowers soil fertility by disrupting C and N biogeochemical cycles (Maxwell, 1995). Primary metal smelters contribute greatly to the vegetation damages due to SO<sub>2</sub> fumigation, soil acidification, and metal contamination (Freedman and Hutchinson, 1980; Cox and Hutchinson, 1981; Marchwinska and Kucharski, 1990). The trace element uptake from contaminated soils and direct deposition of contaminants from the atmosphere onto plant surfaces can lead

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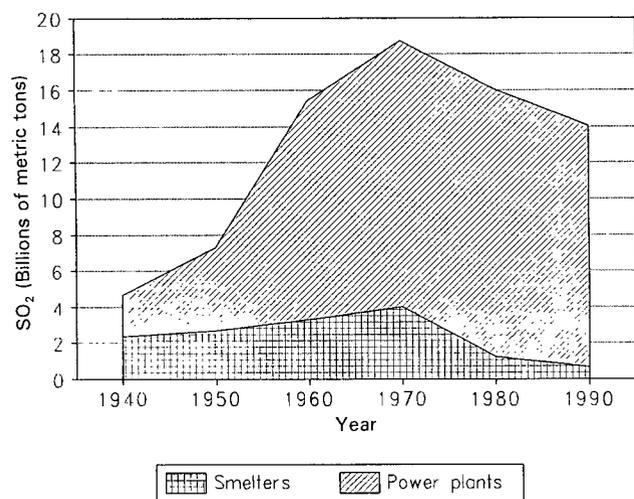


Fig. 1. Estimated global atmospheric emissions of SO<sub>2</sub>.

to the plant contamination by trace elements. Consequently, plant toxicity and the potential for transfer of contaminating elements along the food chain exist (Winterhalder, 1984, 1988; Chaney et al., 1988; Rebele et al., 1993; Dudka et al., 1996b). Soils barren of vegetation are particularly susceptible to erosion, which leads to further damage of the environment in the vicinity of smelters.

The impact of metal ore mining and processing on environmental quality is reviewed with an emphasis on soil contamination by trace elements.

### MINERAL RESERVES, METAL PRODUCTION, EMISSIONS, AND RELATED ENVIRONMENTAL CONCERNS

An estimation of global release of trace elements as contaminants into the environment from mining and smelting may be based on estimates of world production (Table 1). Bowen (1979) has suggested that when the rate of mining of a given element exceeds the natural rate of its cycling by a factor of 10 or more, the element should be considered as a potential contaminant. Accordingly, the potentially hazardous elements to the environment are: Ag, Cd, Cr, Cu, Hg, Mn, Pb, Sb, Sn, and Zn. The available amount of mineral is referred to as a *resource* and a smaller fraction that can be extracted at a profit is known as a *reserve*. Estimates of reserves and resources rely on geological and statistical data (McLaren and Skinner, 1987). The production of mineral reserves depends on their availability, market actions, government regulations, and public opinion. Public opinion is a powerful, but hard to predict, force in determining the world mineral demand. For example, the decline in U use was caused by strongly negative public opinion about the safety of nuclear power (Kesler, 1994).

Assuming the present level of consumption, the reserves of such important elements as Zn and Pb may last for only 10 to 25 yr. Copper is in a 25 to 50 yr category and supplies of Ni will be adequate for 50 to 100 yr (Table 1). The practical significance of these

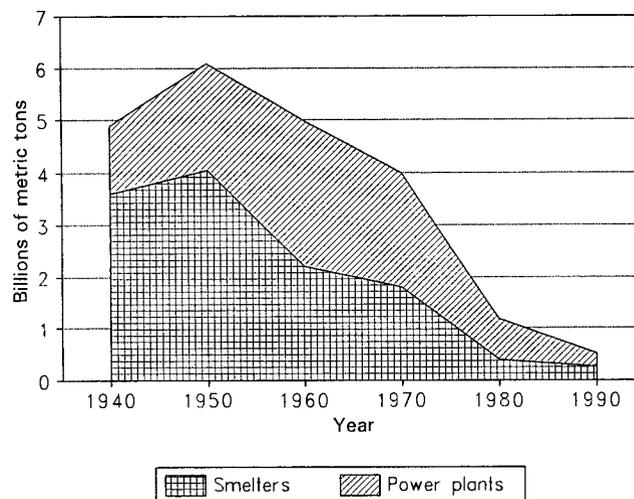


Fig. 2. Estimated global atmospheric emissions of particulate matter.

estimates depends on how long it takes to find reserves and put them into production. Making environmental aspects an integral factor of mineral production prolongs the time to find and develop mineral deposits (Wellmer, 1992) and increases the cost of production. For example, the eight major Cu primary smelters in the USA currently recover about 98% of their SO<sub>2</sub> emissions. Further reduction is possible but at a great cost. A new continuous flush system to be installed at the Cu smelter at Bingham, UT, will recover 99.9% of SO<sub>2</sub> emissions. This system is expected to cost \$880 million, which is about 15% of total expected revenues (Kesler, 1994).

Practically all metals are utilized to some extent in commerce (Minerals Yearbook, 1992). Those produced in major quantities, referred to as *the major metals*, are: Fe, Al, Cu, Pb, Mn, and Zn. The remaining elements are extracted and recovered in smaller quantities and are usually termed *the minor metals*. Among the major metals, the production of Cu, Pb, and Zn causes the greatest environmental concern. In the group of minor metals, As and Cd are the most hazardous because of their geochemical and toxicological properties.

### Copper

Copper occurs naturally in a wide variety of minerals, of which *chalcopyrite* (sulfide) is most common. The biggest deposits are located in the USA, Chile, Canada, Commonwealth of Independent States, Zambia, and Peru (Minerals Yearbook, 1992). The largest market for Cu is electrical motors, generators, and wiring (Fabian, 1986). Most Cu deposits are mined in open pits. These operations generate huge amounts of mine waste and tailings each day, more than mining any other metal (Table 2), and are among the largest earth-moving efforts on the planet (Jolly, 1985). Tailings from Cu-sulfide ore beneficiation amount to as much as 98% of the ore volume and are disposed of in large piles or valley fills (Barney, 1980). Open-pit Cu mines are rarely filled after mining, because they are so large and the ore bodies are not lying flat or near to the surface. Modern mines

**Table 1. Global mineral production, reserves, reserves base, and predicted consumption in 2000. Units are based on element mass unless noted.†**

Element	Mine production	Reserves	Reserve base	Resources	Forecast for year 2000‡	Time until adequate supplies depleted
	Gg yr <sup>-1</sup>			Gg		yr
Ag	13.7	280	420	¶	12	
As§	45	1 000	1 500	11 000	51	10-25
Hg	4.8	130	240	600	14	
Pb	3 200	63 000	130 000	1 400 000	5 000	
Zn	7365	140 000	330 000	1 800 000	11 000	
Cd	20	540	970	6 000	20	25-50
Cu	8 900	310 000	590 000	2 300 000	12 000	
Mn§	18 800	800 000	4 800 000	large	18 000	
Mo	108	5 500	12 000	50 000	130	
Se	1.8	75	130	na	2	
Sn	200	8 000	10 000	na	300	
Co	24.8	4 000	8 800	11 000	30	50-100
Ni	916	47 000	110 000	130 000	1 500	
Sb	60.6	4 200	4 700	5 100	100	
Zr	765	49 000	58 000	>60 000	500	
V	32.1	10 000	27 000	63 000	35	>100
Cr§	12 800	1 400 000	6 800 000	11 000 000	3 750	

† Bowen, 1979; Kesler, 1994; Minerals Yearbook, 1992.

‡ Gg of element.

§ Gg of ore.

¶ Depends on base metal production.

are regraded and revegetated when they are abandoned, as are the waste dumps and tailings piles. This was not true for the older Cu mines, which are a source of metal-rich, acid water and sediments. Environmental concern with Cu production has centered on emissions of SO<sub>2</sub> and easily vaporized trace elements (As, Cd, Hg) from smelters; however, Cu emission is also a serious problem. More than 65% of anthropogenic atmospheric emissions of Cu come from smelting (Fig. 3).

### Lead and Zinc

Lead and Zn are often found together in ore deposits, but the metals have different applications and biological effects. Zinc is a physiologically essential element, whereas Pb has no known positive biological function and creates serious environmental and health hazards. Emissions from Pb smelters and refineries are closely regulated, as are workers' blood Pb levels (WHO, 1989). This adds to the cost of producing Pb, but is necessary to protect both the general public health and the health of Pb industry workers. As a result, primary mine production of Zn has been increasing during the last decade, whereas Pb production has begun to decline (Kesler,

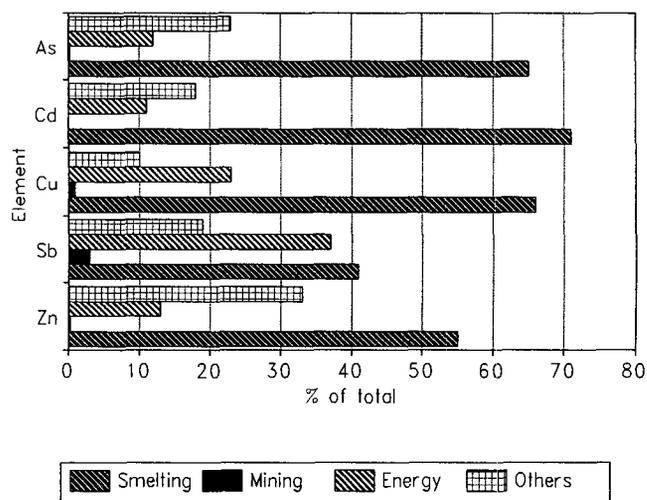
1994). Lead is obtained from *galena* and Zn comes from *sphalerite* (Kesler, 1978). Lead is mined in 47 countries vs. 50 countries for Zn, making them among the most widespread metals in terms of primary production (Minerals Yearbook, 1992). Secondary Pb smelters, which process Pb metal products, are located in 43 countries, but only 21 countries have secondary Zn smelters, reflecting the large recycling of Pb in electric storage batteries (Kesler, 1994). The extensive recycling of Pb results in smelter metal production being substantially higher than mine production.

Lead and Zn processing is an important historical source of environmental contamination with Cd, Pb, Zn, and other elements (Thornton and Abraham, 1984; Asami, 1988a). Metal-rich dispersion zones consisting of galena particles, around the smelter or mine, contain

**Table 2. Estimated cumulative amounts of mine waste and tailings from 1910 through 1981 in the USA in millions of Mg.†**

Mining industry	Tailings	Mine waste	Total waste
		Tg	
Cu	6 900	17 000	23 900
Fe	3 000	8 500	11 500
U	180	2 000	2 180
Mo	500	370	870
Zn	730	70	800
Au	350	400	750
Pb	480	50	530
Ag	50	30	80
Total	12 190	28 420	40 610

† USEPA, 1985.

**Fig. 3. Estimated global atmospheric emissions of selected trace elements from anthropogenic sources; others include manufacturing processes, commercial use (including agricultural uses), and waste incineration (Nriagu and Pacyna, 1988; Nriagu, 1989).**

relatively immobile Pb. Lead oxides and sulfates from flue gases are much more soluble and can be moved to deeper levels in the soil by rainwater. They are also dispersed farther from the source because of their small particle size. Although smelter emissions have decreased significantly in the USA in recent years, most have not achieved the 1978 USEPA limit of  $1.5 \mu\text{g Pb m}^{-3}$ . The cost of full compliance would be about 2.3 U.S. cents  $\text{kg}^{-1}$  of Pb produced, more than can be supported by present profit margins (Isherwood et al., 1988).

The most important market for Pb is in electric storage batteries, which currently account for 80% of the U.S. Pb consumption (Minerals Yearbook, 1992). Zinc is used to make galvanized steel and various Zn alloys (Ohnesorge and Wilhelm, 1991). Three markets in which Pb was the most important a few decades ago now nearly disappeared, are as follows:

1. Use of tetraethyllead as an antiknock additive in gasoline has almost completely ceased in more developed countries, although it continues in some less developed countries (Nriagu, 1990).
2. Use of Pb oxides in interior paint, glass, ceramics, and other chemicals has also declined because of its hazardous characteristics (Ewers and Schlipkoter, 1991).
3. Lead is no longer used in plumbing, a factor that is thought to account for the decline in Pb content of humans over the last few decades (Ewers and Schlipkoter, 1991).

Lead has been so widely used that anthropogenic emissions have greatly outweighed natural releases of this metal. The primary smelting of sulfide ores is the second most important source of anthropogenic Pb after transportation (Table 3). In response to numerous environmental concerns, the U.S. Congress has considered legislation to place a direct tax on Pb production. The proposed legislation would impose a tax of  $\$1.65 \text{ kg}^{-1}$  on primary Pb production, about 250% of the present price of the metal, effectively stopping domestic Pb production within 2 yr (Biviano and Owens, 1992). Because

of the close relation between Pb and other metals, the proposed tax would cut the U.S. production of Zn, Cd, and Bi by >80% and Au production by about 15% (Biviano and Owens, 1992).

### Arsenic

Arsenic is well-known for its toxicity (WHO, 1981). Large doses of As (>100 mg per person) induce acute arsenic poisoning resulting in death (Leonard, 1991). Arsenic is used largely in wood preservatives, herbicides, and insecticides. Because of its wide-spread occurrence with other metals, As is recovered largely as a by-product (Minerals Yearbook, 1992). Arsenic vaporizes at only  $615^\circ\text{C}$  and therefore is released during the roasting of base-metal ores. Older smelters did not capture this vaporized As at scrubbers and are surrounded by zones rich in As. Around the Sudbury Cu-Ni smelters in Ontario, As concentrations in lake sediments deposited during the last few decades are several times higher than in those from times before smelting began (Nriagu and Rayo, 1987).

### Cadmium

Rechargeable batteries with Cd, Ni, Au, or Hg if used in power electric cars would help to resolve the fossil-fuel crisis but Cd is regarded as, potentially, one of the most toxic trace elements in the environment (Adriano, 1986). Cadmium is particularly hazardous because of its easy uptake by plants, its tendency to accumulate in food chain crops, and its persistent nature once it is in the environment. In spite of the environmental concerns, Cd production is not decreasing and consequently as much as 60% of the total input of Cd into air comes from smelting and mining (Fig. 4). The metal is recovered from flue dust during the roasting and sintering of *sphalerite* and in sludge from the electrolytic refining of Zn (Plunkert, 1985). With the production of 1 Mg of Zn, 3 kg of Cd are produced (Ohnesorge and Wilhelm, 1991). Therefore, environmental problems caused by Zn production, at least partly, are attributed to Cd releases to the environment.

Table 3. Estimated global emissions of trace elements to the atmosphere.†

Element	Smelting and refining	Mining	Energy production	Other sources‡	Total anthropogenic	Natural
$\text{Gg yr}^{-1}$						
As	12.3	0.1	2.2	4.3	18.9	12.2
Cd	5.4	-	0.8	1.3	7.5	1.4
Cr	-	-	12.7	17.8	30.5	43.3
Cu	23.2	0.4	8.0	3.6	35.2	56.1
Pb	46.5	2.5	12.7	270.6§	332.3	12.2
Mn	2.5	0.6	12.1	23.0	38.2	316.9
Hg	0.1	-	2.3	1.2	3.6	2.5
Ni	4.0	0.8	42.0	4.8	51.6	29.3
Se	2.2	0.2	3.8	0.1	6.3	10.3
Sb	1.4	0.1	1.3	0.7	3.5	2.6
Sn	1.1	-	3.3	0.8	5.2	-
Tl	-	-	1.1	4.0	5.1	-
V	0.1	-	84.0	1.2	85.3	27.7
Zn	72.0	0.5	16.8	42.5	131.8	44.7

† Nriagu and Pacyna, 1988; Nriagu, 1989.

‡ Manufacturing processes, commercial use (including agricultural uses), and waste incineration.

§ Transportation accounts for 92% emission from other sources.

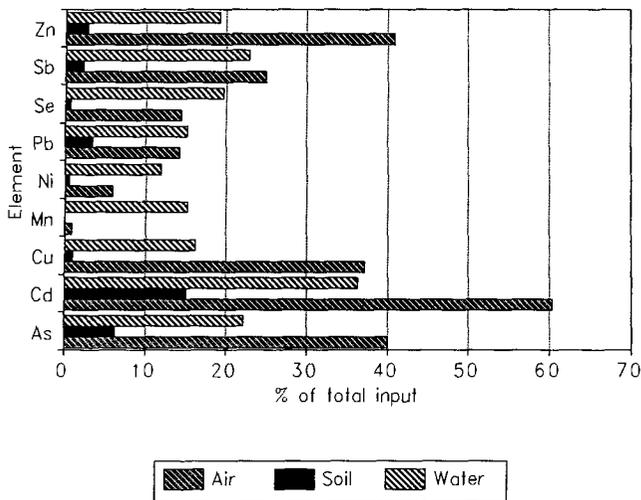


Fig. 4. Proportion of trace element emissions from mining and smelting into air, soil, water (Nriagu and Pacyna, 1988; Nriagu, 1989).

### ENVIRONMENTAL IMPACTS

In the nonfuel mining industry, the valuable portion of the crude ore is a small fraction of the total volume of material that must be handled to obtain it. For example, in the Cu mining industry, the ratio of material handled to units of marketable metal is 420:1 with a typical proportion of metal in ore 0.6% (Minerals Yearbook, 1992). The fact that the materials handled consist largely of waste or unusable materials distinguishes mining industries from many other process industries.

Mining and beneficiation processes generate four categories of large-volume waste: *mine waste* (overburden, barren rocks), *tailings*, *dump heap leach*, and *mine water* (Salomons and Forstner, 1988a,b). The majority of non-fuel ores in the USA are mined by surface techniques (Mineral Yearbook, 1992). Surface mining generates more waste than underground mining (USEPA, 1982). The volume of mine waste as a percentage of the total crude ore ranges from 9 to 27% for underground mines. In surface mining, the amount of waste ranges from 2 to 10 times the total volume of crude ore (Minerals Yearbook, 1992). Tailings are produced from ore beneficiation. The crushed ores are concentrated to release ore particles (value) from the matrix of less valuable rock (Kesler, 1994). Dump leaching, heap leaching, and in situ leaching are the processes used to extract metals from low-grade ore. Sulfuric acid is usually applied in Cu operations. As the liquid percolates through the ore, it leaches out metal. Dump leach piles often cover hundreds of hectares, rise to 60 m, and more and contain tens of millions of Mg of low-grade ore, which becomes waste after leaching. Heap leaching operations are much smaller than dump leach operations and last over a period of months rather than years. The mine water is water that infiltrates a mine and must be removed to facilitate mining. The mine industry wastes in the USA are generally deposited on-site (USEPA, 1985).

Smelting and refining processes produce *gaseous* and *particulate matter* emissions, *waste waters*, and *solid wastes* (slag). These are emitted into ambient air, dis-

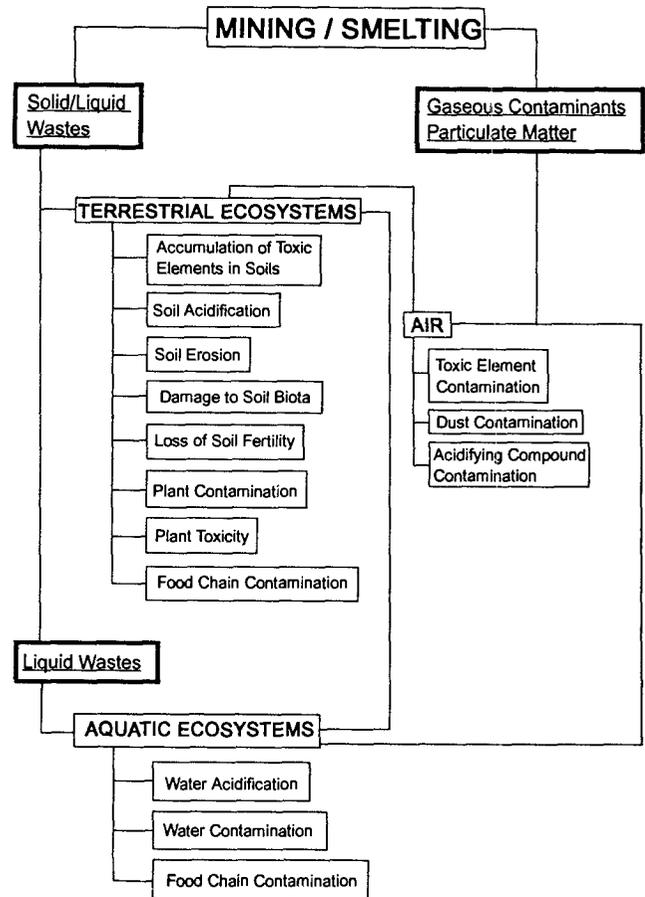


Fig. 5. Schematic diagram of the potential environmental impacts of mining/smelting industries.

charged into water systems, or disposed on land (Fig. 4 and 5). The most noticeable form of contamination from the metal production industry is the discharge of emissions to the atmosphere. Tall stacks discharge pollutants at such heights that the emissions are sufficiently diluted when dispersed into the lower atmosphere to meet air quality requirements (Dobrin and Potvin, 1992). The major solid wastes generated by smelting and refining, which may constitute hazardous waste, include: process wastes, residuals from air pollution control, and waste water treatment systems (Kesler, 1994).

### Soil Degradation

In mining and smelting areas, soils are affected by disposal of mine tailings, acid mine drainage (AMD), and aerial deposition of contaminants from smelters. The exposed soils become acidified and contaminated with trace elements (Fig. 5). The elemental contamination and acidification lower soil fertility and reduce the variety and change the proportion among species of soil biota, and affect the energy budgets of soil fauna (Donker, 1992; Maxwell, 1991, 1995). The negative impact of mining on surrounding lands has largely been related to production of huge amounts of tailings (Table 2). The area of land disturbed by mining in the USA was estimated to be approximately 386 000 ha yr<sup>-1</sup> in 1986 and by 2000 is forecasted to increase to 924 000 ha

yr<sup>-1</sup> (Soni and Vasistha, 1986). According to Donovan et al. (1976) the total accumulated mineral waste in the USA in 1976—including overburden, submarginal ores, milling wastes, and strip-mine spoils—was estimated to be >23 billion Mg and to cover 2 million ha of land.

Abandoned mine tailings have diverse physical, chemical, and ecological properties; however, acid drainage due to sulfide oxidation and elevated levels of trace elements are usually common characteristics of most tailings generated by metal mining (Michelutti and Wiseman, 1995). Also, wind and water erosion and the associated environmental degradation are widespread problems related to tailing material. Leaching of acidifying compounds from tailings leads to an acidification of adjacent soils.

### Acidification

Soils often undergo strong acidification in areas where metal ores are extracted and processed. This process is caused by acidifying compounds emitted from smoke stacks and draining acidic waters from mines and tailings. Mining often exposes sulfide-bearing minerals (pyrite, pyrrhotite, chalcopyrite) to the atmosphere. Metal processing does not remove all pyritic minerals and tailings contain significant sulfide concentrations (Sorensen et al., 1980). A sequence of chemical reactions leads to oxidation of pyrites and production of acid:



The oxidation of ferrous (Fe<sup>2+</sup>) to ferric (Fe<sup>3+</sup>) cations is usually accelerated by the presence of *Thiobacillus ferrooxidans* bacteria (Nordstorm, 1982). The amount of acidity produced by FeS<sub>2</sub> is a function of a many variables—temperature, oxygen supply, concentration of sulfides, initial pH of the surroundings, total Fe concentrations, and bacteria presence (Pugh et al., 1984). Formation of H<sub>2</sub>SO<sub>4</sub> decreases the pH of the tailing environment and adjacent soils, which usually results in increased mobility of metals that are present there.

Also emissions of SO<sub>2</sub> from smelters (Fig. 1) contribute to the soil acidification. For example, the main concern related to Cu production is emission of SO<sub>2</sub> into the atmosphere due to the sulfidic nature of these ores. The amount of S released varies depending on origin of the ore. Sulfur emissions from Cu smelters in Canada during the mid-1980s were about 0.7 Mg Mg<sup>-1</sup> of Cu produced vs. 0.11 in the USA, and 0.03 in Western Europe (Kesler, 1994). In the Sudbury Cu–Ni mining and smelting basin (Ontario, Canada) soils are strongly acidified (pH in range 2.0–7.5) as a result of SO<sub>2</sub> fumigation from two active primary Cu–Ni smelters (Dudka et al., 1995b). Metals present in the acidic soils become easily mobile and available for plants, and therefore, their leaching, plant uptake, and runoff from soils increase. Sudbury soils had the concentrations of mobile Al as high as 100 mg kg<sup>-1</sup>, which is apparently one of the factors of plant toxicity of these soils. Also high concentrations of exchangeable Cu and Ni were detected in these soils (Adamo et al., 1996; Dudka et al., 1996b). Extreme measurements were about 90 and 300

mg kg<sup>-1</sup> (dry wt.) of Cu and Ni, respectively. There was a good linear relationship between the level of the exchangeable metals in the soils and their concentrations in birch (*Betula papyrifera* Marsch) leaves and twigs. Prolonged soil acidification decreases base saturation of soils, reduces humus content, and alters mineral composition of soil clay fraction (Motowicka-Terelak and Dudka, 1991). The rate of this process depends on the level of SO<sub>2</sub> deposition, soil mineral composition, and particle-size distribution of soils (Motowicka-Terelak and Dudka, 1991).

### Soil Biology

The components of soil biota are diverse. They include bacteria, fungi, algae, and soil fauna. Soil microorganisms contribute to structural integrity of the soils, to the changes in forms of plant nutrients through the mediation of various biogeochemical cycles involved in nutrient recycling, and can act in several ways to alleviate adverse properties in the soils, thus improving conditions for growth of higher plants (soil fertility). The abundance and diversity of soil microorganisms are generally reduced by smelter emissions and acidic conditions (Maxwell, 1995). The soil flora of contaminated sites in Sudbury was characterized by a low diversity of chlorophytes, one or two diatom species, and a notable absence of cyanobacteria (Maxwell, 1991). Similar results have been reported for soils subject to emissions from various metallurgical plants in Russia (Shtina et al., 1985). These authors found that cyanobacteria were absent from soils within a radius of 20 km. The absence of cyanobacteria from Sudbury soils is apparently the result of strong soil acidification. The chlorophyte-dominated flora of the Sudbury barrens is characteristic of acid soils. *Chlamydomonas acidopholia* is ubiquitous in these soils. This acid-tolerant algae was also found in strip-mine ponds of Ohio with pH of 3.3 (Rhodes, 1981). According to Twiss (1990) and Hutchinson et al. (1981), green algae isolated from Sudbury soils were not only acid-tolerant but also metal-tolerant.

### Trace Metals

Mining and smelting are not the main source of global trace metal input into soil (Table 4). Other sources like discarded manufactured products, coal ashes, agriculture, and transportation take a lead (Table 4); however, strong soil contamination with trace metals in mining/smelting regions has been reported locally in numerous case studies (Tables 5–9). It is a great challenge to analyze and synthesize the soil data reported by various authors. Sampling techniques applied are not consistent, ranging from subjective (and obviously biased) approaches to random or systematic to designs based on geostatistical methods. In many cases, the number of collected samples is not representative to infer the conclusions about studied areas. One or few extremely high observations, which may be a result of sampling or analytical error, make a very broad range of concentrations, which is the basis for conclusions about strong contamination of an area. Besides, analytical methods produce

Table 4. Estimated global inputs of trace elements into soils.†

Element	Discarded manufactured products	Atmospheric fallout‡	Gg yr <sup>-1</sup>			Total
			Coal ashes	Agriculture§	Other sources¶	
Sb	2.4	2.5 (0.6)	12.0	13.0	3.8	26
As	38.0	13.0 (5.2)	22.0	22.2	2.5	82
Cd	1.2	5.3 (3.3)	7.2	4.8	5.5	22
Cr	458.0	22.0	298.0	116.0	38.0	898
Cu	592.0	25.0 (9.4)	214.0	74.0	71.0	971
Pb	292.0	232.0 (33.1)	144.0	30.2	62.1	959
Mn	300.0	27.0 (0.2)	1076.0	200.0	95.7	1669
Hg	0.7	2.5 (0.05)	2.6	2.5	1.7	8.3
Ni	19.0	24.0 (1.4)	168.0	80.4	35.8	294
Se	0.2	2.0 (0.3)	32.0	10.1	2.1	41
V	1.7	60.0 (0.03)	39.0	6.4	7.1	128
Zn	465.0	92.0 (37.6)	298.0	150.0	149.0	1322

† Nriagu and Pacyna, 1988; Nriagu, 1990.

‡ Number in brackets represent amounts of elements from mining and smelting.

§ Includes agricultural and animal wastes, fertilizers, and peat.

¶ Includes logging and wood wastes, urban refuse, organic wastes, and solid waste from metal fabrication.

incomparable results. In many papers there is no information about quality control of the methods applied; therefore, a reader does not know how accurate and precise the reported measurements were. Also, statistical analysis of the data varies greatly. In many cases only ranges of concentrations are given or merely single observations are listed. Hardly any attempt was made to estimate what percentage of samples, therefore proportion of studied area (assuming that sampling was representative), had elemental concentrations in particular classes of values. Authors usually do not report the type of distribution of element concentrations. They automatically assume normal distribution and report the measurement of central tendency and variability as arithmetic means and standard deviations; however, comparison of reported means and standard deviations indicates that the distributions were far from normal and rather geometric means and geometric deviations should have been used to summarize results. The use of proper form of mean is important because when data sets are highly positively skewed, an arithmetic mean is highly overestimated.

Despite these reservations, there is much evidence that Pb and Zn production leads usually to the most severe soil contamination (Table 5–9), not only by Pb

and Zn but also by Ag, As, Cd, Cu, and Ni. Soil contamination by metals from mining and smelting was comprehensively studied in Japan, the UK, and Poland. Also the fate and effects of the emission of contaminants from the Cu–Ni smelters of Sudbury, northern Ontario, Canada, into the surrounding environment are believed to be better documented than those of any other smelter in the world (Asami, 1988a).

Although the UK is no longer an important producer of Pb and Zn ores, the several case studies show soil contamination by these metals (Angelone and Bini, 1992). The most extensive source of metal contamination in Great Britain is metalliferous mining, which commenced in Roman times and had been operating until the end of the 19th century (Colbourn and Thornton, 1978; Thornton and Abraham, 1984). It has been estimated that >4000 km<sup>2</sup> of agricultural land in England and Wales is contaminated with one or more metals because of these historical operations. It is interesting to note that despite of extremely high concentrations of Cd, Pb, and Zn detected in the soils reclaimed from an old mine at Shipham in southwestern England, little of these metals has been transferred into pasture plants (Thornton et al., 1980; Matthews and Thornton, 1982). This observation is a good evidence of the effect of metal forms on plant availability (Kuo et al., 1983). High metal concentrations in the Shipham soils result from mineralization, not emissions of contaminants; there-

Table 5. Cadmium concentrations in surface soils in the vicinity of mines and smelters.

Range	Mean	Source of contamination	References
mg kg <sup>-1</sup> dry wt.			
24–440	159	Zn–Pb mining	Matthews and Thornton, 1982
1.0–3.9	1.8	Old mine	Colbourn and Thornton, 1978
0.4–540	6.1	Old mine	Davies and Roberts, 1978
1.2–94	18	Old mine	Davies and White, 1981
0.3–102	2.5	Zn–Pb smelting	Dudka et al., 1995a
0.1–10	4.7	Cu–Ni mining and smelting	Dudka et al., 1995b
0.4–132	3.2	Mining/smelting	Dudka and Sajdak, 1992
1.8–9.2	3.9	Zn–Pb mining and smelting	Asami, 1988a
11–1781	–	Zn smelting	Seokart et al., 1983
3–750	–	Zn smelting	Jordan and Lechevalier, 1975
5–14	–	Cu smelting	Kuo et al., 1983
0.02–10.9	1.1	Cu smelting	Roszyk and Szerszen, 1988a
	12	Zn–Pb smelting	Rauta et al., 1987

Normal Cd soil level: 0.1–1

Table 6. Copper concentrations in surface soils in the vicinity of mines and smelters.

Range	Mean	Source of contamination	Reference
mg kg <sup>-1</sup> dry wt.			
11–1 890	116.0	Ni–Cu mining and smelting	Dudka et al., 1995b
60–3 700	–	Ni–Cu mining and smelting	Freedman and Hutchinson, 1980
76–9 700	–	Ni–Cu mining and smelting	Hazlett et al., 1983
48–15 000	–	Cu smelting	Rebele et al., 1993
30–3 280	460.5	Cu smelting	Roszyk and Szerszen, 1988a,b
40–1 989	–	Zn smelting	Seokart et al., 1983
24–9 618	–	Cu smelting	Jordan and Lechevalier, 1975

Normal Cu soil level: 2–20

**Table 7. Lead concentrations in surface soils in vicinity of mines and smelters.**

Range	Mean	Source of contamination	Reference
mg kg <sup>-1</sup> dry wt.			
6-7 100	160	Zn-Pb smelting	Dudka et al., 1995b
8-710	85	Cu-Pb-Bi smelting	Asami et al., 1990
49-371	150	Ag-Cu-Pb-Zn mining smelting	Asami, 1988a
30-18 400	218.6	Cu smelting	Roszyk and Szerszen, 1988a
28-4 226	-	Cu smelting	Rebele et al., 1993
4-8 200	102	Mining/smelting	Dudka and Sajdak, 1992
475-7 800	3 829	Zn-Pb mining	Matthews and Thornton, 1982
280-448	-	Zn-Pb mining and smelting	Davies and Roberts, 1978
38-14 910	1 759	Pb mining	Davies and White, 1981
190-3 000	-	Old mine	Colbourn and Thornton, 1978
189-14 000	-	Zn smelting	Scokart et al., 1983
92-2 580	862	Zn-Pb smelter	Nwankwo and Elinder, 1979
Normal Pb soil level: 5-30			

fore, the metals are strongly fixed to primary soil minerals.

The published information on soil contamination by Cu, Ni, and other elements in the Sudbury area, Ontario, Canada, consists mainly of sets of scatter point-data (e.g., Rutherford and Bray, 1981; Freedman and Hutchinson, 1980; Cox and Hutchinson, 1981; Hazlett et al., 1983; Taylor and Crowder, 1983; Negustanti and McIlveen, 1990; Gundermann and Hutchinson, 1993). An exception is a recent study (Dudka et al., 1995b), which provided enough data to produce maps of Cu and Ni occurrences in surface soils of the Sudbury area. The Sudbury region has a history of mining and smelting of sulfide ores lasting >100 yr. These soils contain high levels of total Cu, Ni, Cd, Co, Cr, and S. The concentrations of Cu and Ni, which are the main contaminants for Sudbury's soils, often exceed 1000 mg kg<sup>-1</sup> (Negustanti and McIlveen, 1990; Dudka et al., 1995b). A quantitative estimation of the degree of element contamination requires the reference (background) values against which the current concentrations are being compared. The Ontario Ministry of Environment established a set of guidelines (Co, 25; Cu and Ni, 60; S, 1000 mg kg<sup>-1</sup>, dry wt.) to indicate the concentrations of elements that are above background limits (Negustanti and McIlveen, 1990). The upper limits of these guidelines represent the expected maximum concentrations of elements in nonagricultural surface soils. Concentrations of Cd, Co, Cu, Cr, Fe, Mn, Ni, Zn, and S in the

**Table 8. Zinc concentrations in surface soils in vicinity of mines and smelters.**

Range	Mean	Source of contamination	References
— mg kg <sup>-1</sup> dry wt. —			
11-10 500	205	Zn-Pb smelting	Dudka et al., 1995a
49-554	-	Cu smelting	Rebele et al., 1993
238-472	349	Zn-Pb mining and smelting	Thornton et al., 1980
7-910	-	Cu smelting	Drozd et al., 1984
2040-50 000	14 970	Zn-Pb mining	Matthews and Thornton, 1982
10-49 390	728	Zn-Pb mining	Davies and Roberts, 1978
11-641	96	Pb mining	Davies and White, 1981
400-4 245	-	Zn-Pb mining	Letunova and Krivitskiy, 1979
1340-180 000	-	Zn smelting	Scokart et al., 1983
180-3 500	1 050	Zn-Pb smelting	Nwankwo et al., 1979
430-1 370	-	Cu smelting	Kuo et al., 1983
110-60 700	-	Zn smelting	Jordan and Lechevalier, 1975
	554	Ag-Cu-Pb-Zn mining and smelting	Asami, 1988a
Normal Zn soil levels: 15-100			

studied soils exceeded the established normal concentrations. Only about 25 and 38% of the soil samples collected had normal concentrations of Cu and Ni, respectively.

There is a well-established opinion in the literature that the retention time for trace metals in soils, measured as a time span needed to reduce the element concentration by half, is from hundreds to thousands of years (Kabata-Pendias and Pendias, 1992). Cadmium, Ni, and Zn are considered to have the shortest retention time, Cu medium, and Pb and Cr the longest retention time under similar environmental conditions. Data on Pb migration in forest soils question this assessment (Miller and Friedland, 1994). These authors found significant reduction in Pb concentrations in the forest floor of the northern USA in 10 yr after use of leaded gasoline was prohibited in North America. Also in Sudbury, substantial differences in the soil concentrations of several elements can be detected within the period of only 10 to 20 yr. There are not conclusive data yet to precisely explain the mechanism of the metal concentration reduction in the Sudbury soils, although it seems probable that two factors are involved (Dudka et al., 1995b): (i) leaching and runoff of metals into water systems; and (ii) removing fine soil particles, enriched with metals, by erosion.

However, it should be pointed out that erosion does not change the metal concentrations in the soil material.

**Table 9. Element concentrations in surface soils in vicinity of mines and smelters.**

Element	Range	Mean	Normal level	Source of contamination	References
mg kg <sup>-1</sup> dry wt.					
As	0.3-145	10.2	1-15	Cu smelting	Roszyk and Szerszen, 1988a
Bi	0.4-122	4.2	0.3	Cu-Pb-Bi smelting	Asami et al., 1992
Cr	30-4560	-	4-80	Cr smelting (smelter slag)	Asami, 1988b
Co	1.3-38.3	5.8	1-10	Cu smelting	Roszyk and Szerszen, 1988b
Hg	0.01-15.1	0.9	0.01-0.5	Cu smelting	Roszyk and Szerszen, 1988b
In	0.02-1.9	-	0.02-0.08	Zn-Pb mining	Asami et al., 1990
Ni	5-2150	105	2-30	Ni-Cu mining and smelting	Dudka et al., 1995a
	100-3000	-	-	Ni-Cu mining and smelting	Freedman and Hutchinson, 1980
	160-12300	-	-	Ni-Cu mining and smelting	Hazlett et al., 1983
	3-450	36	-	Cu smelting	Roszyk and Szerszen, 1988b
Sb	0.6-37	3.2	0.4	Cu-Pb-Bi smelting	Asami et al., 1990

The result of erosion is translocation of the metal-contaminated material and its dispersion over a bigger area. Leaching and washing of the metals from the studied soils is apparently facilitated by the low soil pH. An acidic reaction is the most important single factor enhancing soil metal mobility. Erosion can be of importance for the local reduction of metal concentrations, especially in barren and semibarren areas.

Upper Silesia (southern Poland) is a heavily industrialized and agricultural area. The industries present in Upper Silesia include: black coal mines, metal ore mines, Pb-Zn smelters, Fe smelters, coal-fired power plants, and many others. Metal contamination there is mainly caused by two active Zn-Pb smelters (Gzyl, 1990; Pawlowski, 1990; Marchwinska and Kucharski, 1990; Dudka et al., 1995a). The arable land covers about 321 000 ha of predominantly medium quality soils, which accounts for 48% of the total area (Dudka et al. 1995a). Emission of trace metals in this region in 1990 was ( $\text{Mg yr}^{-1}$ ): Pb, 235; Cr, 98; As, 7; and Cd, 4 (Smal and Salomons, 1991). At some areas in Upper Silesia the deposition of Cd and Pb exceeds up to 10 times the proposed limits, which are  $2.75 \text{ kg km}^{-2}$  for Cd and  $182.5 \text{ kg km}^{-2}$  for Pb (Smal and Salomons, 1991). The arable soils in Upper Silesia are contaminated with Cd, Pb, and Zn; occasionally high concentrations of Cu have been detected (Gzyl, 1990; Dudka and Sajdak, 1992; Dudka et al., 1995a). The most serious contamination occurs at the vicinity of two active Pb-Zn smelters; however, other areas of the region are affected by a long-range transport of contaminants. The ranges of concentrations ( $\text{mg kg}^{-1}$ , dry wt.) of main metallic contaminants are broad: Cd, 0.1 to 143.0; Pb, 4 to 8200; Zn, 5 to 13 250. The geometric means of heavy metal concentrations in the soils of the region are (in  $\text{mg kg}^{-1}$ ): Cd, 2.5; Pb, 160.4; and Zn, 204.6 (Dudka and Sajdak, 1992; Dudka et al., 1995a). It is estimated that only 10% of the arable land has natural concentrations of the heavy metals, 30% of the soils is slightly contaminated and 60% is medium and heavily contaminated (Dudka and Sajdak, 1992).

Japan is well known for environmental contamination, including soil contamination from mining and smelting (Asami, 1988a,b). There are several reasons why severe contamination has occurred in that country. Japan has a very high population density, a large number of metal mines, and the country experienced a rapid economic development after World War II. The abrupt increase in metal production spread metals from mines, smelters, and metal processing factories. According to Asami (1988a), mining/smelting industries are the main source of Cd contamination of Japanese soils. There are also areas with soils contaminated by As, Be, Bi, Ga, Zn, Cr, and Pb from mining/smelting in Japan (Asami, 1988b; Asami et al., 1990).

### Trace Metal Contamination of Terrestrial Plants

Uptake from contaminated soils and deposition of contaminants from the atmosphere onto plant surfaces are sources of elevated levels of trace elements in terrestrial plants grown in mining/smelting areas. The crops in Upper Silesia (southern Poland) had elevated concen-

trations of Cd, Pb, and Zn, apparently as a result of emissions from two Pb-Zn smelters active in the area; however, the metal levels were not extremely high. Geometric means of Cd content of potato (*Solanum tuberosum* L.) tubers and cereal grain (*Triticum vulgare* L., *Secale cereale* L., *triticale*, *Hordeum vulgare* L., and *Avena sativa* L.) were 0.52 and  $0.22 \text{ mg kg}^{-1}$ , respectively (Dudka et al., 1995a); however, risk assessment of the food chain contamination revealed that no excessive dietary intake of Cd is expected when Cd concentrations in grain and potato tubers grown on the industrially contaminated soil are not higher than 0.6 and  $1.0 \text{ mg kg}^{-1}$ , respectively (Dudka et al., 1996a).

The extensive area of Sudbury soils remains barren or semibarren of vegetation as a result of severe environmental conditions (Winterhalder, 1984). Following environmental improvement in the Sudbury area, several plant species including tufted grass (*Deschampsia cespitosa*), tickle grass (*Agrostis scabra* Willd.), redtop (*A. gigantea* Roth), and dwarf birch (*Betula pumila* var. *glandulifera*) have colonized barren sites (Amiro and Courtin, 1981; Winterhalder, 1984); however, recolonization was confined to relatively favorable sites by species that have evolved metal tolerance (Cox and Hutchinson, 1981).

### HEALTH RISK

The evaluation of environmental health impacts of mining and smelting is a very difficult task, because of the complexity of factors that should be submitted to assessment as well as lack of uniform methodology, which may lead to conflicting results (Baker and Bowers, 1988; Dutkiewicz, 1993). A good example of an extremely complex environmental situation is Upper Silesia, in southern Poland. This region, with a population of 2 million, is a region of numerous industries—ferrous and nonferrous mining and smelting, hard coal mines, power plants, and others (Dudka et al., 1995a). Upper Silesia frequently experiences peak ambient  $\text{SO}_2$  concentrations of  $>300 \mu\text{g m}^{-3}$  and average values of more than  $60 \mu\text{g m}^{-3}$ , which is the level when human health can be endangered (Kabala, 1989). In addition, ambient concentrations of aerosols in many sites of Upper Silesia have ambient concentrations of acid aerosols (mainly sulfates) that are high enough to cause a significant deterioration of lung function (Jedrychowski and Krzyzanowski, 1989). Furthermore, it has been estimated that because of the heavy load of air pollutants and stagnant air masses common in many cities of Upper Silesia, the oxygen content of the air can decrease by as much as 20% (Szpunar et al. 1990). This can induce a health hazard to heart patients and asthmatics. The hazardous conditions prevailing in Upper Silesia are believed to be responsible for 15% higher circulatory problems, 30% more cancer cases, and 47% greater respiratory illnesses observed in residents of this region compared with those in the rest of Poland (Kabala, 1985; Szpunar et al., 1990). The incidence of these diseases can be related to the overall deterioration of environmental quality in Upper Silesia caused by emissions from various industries, including metal ore mining and pro-

cessing, and it is impractical to quantify the relative contribution of particular offenders.

Nevertheless, in some situations it is evident that harmful health effects were caused by emissions from mining/smeltering industries. The case studies performed in the vicinity of the Pb mine and Orzel Bialy smelter plant (the northern Upper Silesia, Poland) revealed that the blood Pb (Pb-B) concentrations in exposed children were considerably higher than in a reference group (Dutkiewicz et al., 1993). In 26% of 7 to 8 yr old children and 11% of 14 to 15 yr old children, the Pb-B concentrations exceeded the *assumed* safe limit value of 200  $\mu\text{g L}^{-1}$ . In 54% of the younger children and in 34% of the older children the Pb-B concentrations exceeded the reference value of 140  $\mu\text{g L}^{-1}$  for children from noncontaminated parts of Poland (Dutkiewicz et al., 1993). The Pb-B concentrations of 400 to 500  $\mu\text{g L}^{-1}$  may cause perturbation in hemoglobin synthesis, high coproporphyrin levels, increased level of erythrocyte protoporphyrin, and neurological impairment, leading to decrease of intelligence quotient (IQ). Dutkiewicz et al. (1993) concluded that the existing exposure of children combined with contamination of the environment with Pb compounds from long-term mining and smelting of nonferrous ores creates serious health risks in the investigated region.

Comprehensive studies were conducted to evaluate the state of health and the degree of exposure to heavy metals of children living in the area of the Zn smelting works in Miasteczko Slaskie (Hager-Malecka, 1991). The smelter emits mainly Pb, Cd, and Zn, causing contamination of air, soil, and food products. The surroundings of the smelter are probably the most contaminated areas in Upper Silesia, by virtually any standards. More than 50% of 7 to 10 yr old children had Pb-B higher than the *assumed* upper safe limit of 250  $\mu\text{g L}^{-1}$ , and in 21% of these children Pb-B exceeded 300  $\mu\text{g L}^{-1}$ . A substantially lower percentage of younger and older children had elevated Pb-B. Psychological examination revealed a correlation between Pb concentrations and IQ, but IQ was within the normal range in the studied children. No correlation was found between Pb-B and the distance of the place of residence from the Zn smelter. Nor was a correlation found between Pb and Zn concentrations in blood and the consumption of vegetables grown in the contaminated areas. There were no distinct clinical signs of chronic Pb intoxication in children living around Zn smelter at Miasteczko Slaskie. No Zn excess was found in any of the children, but serious Zn deficits were discovered in a large number of the examined children.

Food of plant origin is a main source of Cd intake by modern society (Dudka and Miller, 1997). Much concern has been generated about soil Cd contamination because farmers in Japan, where Cd soil contamination has occurred after field irrigation by mine waters, actually experienced adverse health effects from soil Cd (Asami, 1991). In these areas in Japan where mine waters form Pb-Zn mine-polluted paddy fields, Cd concentrations increased in rice (*Oryza sativa* L.) grain without any change in grain Zn content (Chaney and Ryan, 1994). A combination of low Ca, low dairy product consumption, and highly increased level of Cd but

low levels of Ca, Zn, and Fe in rice, promoted Cd absorption and caused osteomalacia in women in Japan. Dudka et al. (1996a) conducted a long-term field experiment to evaluate transfer of heavy metals from soils contaminated by Pb-Zn smelter flue dust to crop plants grown in rotation. Risk assessment analysis (Dudka et al., 1996a), using empirical data from this experiment, did not indicate any excessive dietary intake of Cd by a general population when Cd concentrations in cereals and potatoes were 0.6 and 1.0  $\text{mg kg}^{-1}$  (the Cd concentrations similar to those in rice from the polluted paddy fields in Japan), respectively. Based on the results of risk analysis and the saturation model of soil-plant metal relationship, it was concluded that a Cd concentration in soil of 30  $\text{mg kg}^{-1}$  is still safe to the food chain, providing that soil pH is in a neutral range. Further the authors stated that Pb and Zn do not pose any risk to the food chain (Dudka et al., 1996a). A study in England showed that soil averaging  $>100 \text{ mg kg}^{-1}$  of Cd caused no human disease to long-term gardening residents; here Zn accompanied the Cd in the gardens (from mine wastes that were dispersed several centuries before housing was constructed), and moved to edible tissues of the garden crops (Chaney and Ryan, 1994). Similar situation of a village being established on mine/smelter waste after World War II, and villagers growing gardens on the contaminated soils were found in Stolberg, Germany (Ewers et al., 1993), and in Palmerton, PA, USA (ATSDR, 1994). The epidemiological studies on long-term exposure to Cd in these communities found no evidence of Cd-induced renal tubular dysfunction problem in any of the tested persons, even those who might have had high enough Cd ingestion to create a problem. Therefore, it seems unlikely that under a Western diet intake of Cd with foods from industrially contaminated land would cause health problems (Dudka et al., 1996a; Chaney and Ryan, 1994). On the other hand, in China several cases similar to the rice poisoning in Japan were observed (Chaney and Ryan, 1995).

## SUMMARY AND CONCLUSIONS

Mining and smelting of metals cause degradation of the environmental quality. Copper mining is an especially a heavy burden on the environment because it generates more than half of all mine wastes and tailings produced in metal mining. Although mines are classified on the basis of their predominant products, they produce large quantities of other elements as coproducts. As a result, metal ore processing usually leads to multi-elemental contamination of the environment. Despite the steady reduction of  $\text{SO}_2$  emissions since the 1970s and particulate matter emissions since the 1950s, smelting is still a significant source of gaseous and particulate contaminants. Copper smelting is regarded as one of the major manmade sources of  $\text{SO}_2$ , which contributes to acid rain. The gaseous, dust, liquid, and solid wastes discharged into the environment from mines and smelters cause soil and water acidification, air, water, soil, and plant contamination by trace elements, deterioration of soil biology and fertility, and soil erosion. Primary metal smelters are the main sources of atmospheric emissions of As, Cu, Cd, Sb, and Zn on a global scale and they

contribute largely to overall emissions of Cr, Pb, Se, and Ni. All known metals have applications in industry, technology, and everyday uses, and the number of their commercial uses has been accelerating. Each industrial process of metal manufacturing generates some kind of waste that is discharged into the environment. Unlike organic chemicals, the metals are nondegradable, very persistent, and their build-up in the environment continues. Quantitative evaluation of environmental health impacts of mining and smelting is a difficult task due to the complexity of interacting factors, thus leading often to conflicting results; however, it is possible that the current levels of metals in the vicinity of many mining and smelting operations are high enough to constitute a hazard to human health. Based on the risk assessment it seems *unlikely* that food crops grown in soils contaminated by smelting are a source of excessive intake of Cd under the western diet.

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