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# ARSENIC IN THE GOLD AND BASE-METAL MINING INDUSTRY

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# ARSENIC IN THE GOLD AND BASE-METAL MINING INDUSTRY

by Kristina Ringwood

## ABSTRACT

Heavy metal contamination of the environment is an issue of current and growing public concern with respect to human health. The gold and base-metal mining industry is a significant source of heavy metals, including *arsenic*, in the environment. As a responsible land manager, the mining industry must develop better strategies and technology for managing these metals, otherwise the loss of public trust will make future access to land for exploration and mining more difficult. The mining industry must learn to anticipate the future concerns of the public for the environment and solve these problems itself, or otherwise be forced to do so by increased regulation and government control.

Arsenic is an element historically known for its toxicity, and certain species are now recognised as carcinogens. Government regulating bodies have accordingly proposed significant reductions in the current maximum permissible concentrations for arsenic in the environment. New and retrospective legislation requiring more stringent environmental controls will therefore result in substantially increased costs for the mining industry. New strategic initiatives by the industry involving investigation of contaminant processes through well-researched case-studies would be advantageous because they should influence the formulation of new legislation.

This paper collates and analyses current technical and policy information for the management of arsenic within the gold and base-metal industry and information on its occurrence, toxicity, environmental geochemistry, emissions to the environment, sources, immobilisation technologies and regulation. In addition, the specific issues concerning arsenic contamination and management in the mining industry are discussed.

Exposure to arsenic is unavoidable, as arsenic is a ubiquitous and naturally occurring, toxic element in the earth's crust. The degree of toxicity of arsenic is dependent on its chemical form and speciation; *inorganic* arsenic compounds are more toxic than *organic* arsenic compounds, and the reduced species of arsenic, *As(III)*, is reported to be 25 to 60 times more toxic than the oxidised species, *As(V)*. Furthermore, *As(III)* is intrinsically more mobile. The controls on arsenic speciation

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and mobility in the environment are discussed as these are important in the design of closure options at waste sites. These controls include Eh, pH, the presence of iron oxide and clays, soil texture and organic matter.

Arsenic is a waste product from the processing of a variety of ores, including copper, gold, nickel, lead and zinc. Smelters, tailings dams, mullock stockpiles and disposal sites constitute sources for potential escape of arsenic into surrounding groundwater and surface water. In addition to the existing problems of arsenic wastes there will be an increase in the future production of arsenic as industry begins to process more complex sulphide ores, such as low grade gold associated with arsenopyrite and nickel ores with high arsenic contents. Moreover, there will be a continuing increase in the global cycling of arsenic resulting from the progressive industrialisation of developing nations.

Arsenic is relatively soluble in water, with solubility increasing with pH at both temperature extremes. Therefore, arsenic wastes cannot be disposed of in areas if it is likely to come into contact with water. Arsenic waste thus requires immobilisation or treatment to remove arsenic from the waste stream. This review describes the current methods available for the treatment of arsenic wastes for subsequent immobilisation and disposal in an environmentally responsible manner.

Research at the Australian Nuclear Science and Technology Organisation (ANSTO) has developed a process which appears to be a most promising technology in terms of long-term stability of mine wastes and the removal of arsenic from contaminated waters. The iron arsenate adsorption method requires the oxidation of As(III) to As(V) prior to the adsorption process. Ongoing work at ANSTO is researching the effectiveness of photolytic oxidation of As(III) to As(V). Recent research of mine waste waters in Victoria has found the exposure of arsenic-bearing water to sunlight is very effective in oxidising As(III). This process would remove the need for chemical oxidants for the oxidation process, thereby decreasing the cost of the iron arsenate adsorption technology.

Concern over the contamination of the environment arises because of the high toxicity of arsenic, particularly in reduced environments such as groundwater where the mobility of aqueous species are increased. Moreover, arsenic is a carcinogen and many governments are in the process of reviewing safety limits for the element, especially in groundwater used as a source of drinking water. The increased controls on emission, disposal and international trade in hazardous wastes (such as the Basel Convention, see Appendix 1) will result in higher costs to the mining industry for managing its wastes. For this reason, contamination of land and water by arsenic requires careful management and quantitative documentation.

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## ARSENIC IN THE GOLD AND BASE-METAL MINING INDUSTRY

### 1. Introduction

This paper presents a review of arsenic, including its occurrence, sources, toxicity, geochemistry, emissions, disposal methods and regulation. This information provides a comprehensive framework in which to address the various issues associated with arsenic contamination in the gold and base-metal mining industry. In addition, this review highlights the limitations of our current knowledge concerning arsenic cycling, toxicity and geochemistry. The evaluation of current practices and future directions for regulation of arsenic emissions is presented. Anticipation of more stringent regulation for heavy metal emissions is essential for the mining industry, in order to reduce future costs of emission control and to avoid retrospective liability for contaminated sites.

### 2. Occurrence

Arsenic is a naturally occurring, toxic element in sediments and waters of the earth's crust, with average concentrations shown in Table 1 (Whitacre and Pearse, 1974). Arsenic is a major constituent of over 245 different minerals (Kipling, 1977) including arsenopyrite ( $\text{FeAsS}$ ), one of the most common minerals in gold and base metal deposits. A selection of arsenic-bearing minerals is shown in Table 2 and includes examples of the type of ore deposit the minerals occur in and the average arsenic concentration of these deposits. Arsenic is a metalloid but is commonly grouped with the heavy metals because of its similar toxic properties and biogeochemical cycling. Arsenic occurs in the environment as native element, alloys, arsenides, sulphides and oxides (Piret and Melin, 1989; United States Environmental Protection Agency (EPA), 1992).

Table 1: Average arsenic concentrations in the environment (Whitacre and Pearse, 1974).

<u>Arsenic</u>	<u>Average Concentrations (ppm)</u>
soils/sediments	1-50
sulphidic sediments	100-300
igneous rocks	1.5
metamorphic rocks	0.4-18
mineralised zones	up to 4% in copper ores
ocean water	0.002-0.003
groundwater (U.S.A.)	<0.001
air	$7.7 \times 10^{-6}$
marine plants	30
terrestrial plants	0.2
marine animals	0.005
terrestrial animals	<0.02

Table 2: Common arsenic-bearing minerals and ore associations

(Whitacre and Pearse, 1974; Kipling, 1977; EPA, 1992)

<u>Mineral</u>	<u>Formula</u>	<u>Occurrence</u>	<u>% As Average</u>
Arsenopyrite	FeAsS	lode gold e.g. Bendigo, Stawell	0.5
		Cu sulphide	4.0
		Sn e.g. Renison	0.2
Cobaltite	CoAsS	Ni, Cu and Zn ores	2.5
Enargite	Cu <sub>3</sub> AsS <sub>4</sub>	vein and replacement, U.S., Chile, Philippines	0.1
Loellingite	FeAs <sub>2</sub>	mesothermal veins	-
Niccolite	NiAs	e.g. Kambalda	0.5
Orpiment	As <sub>2</sub> S <sub>3</sub>	hydrothermal veins	2.0
Tennantite	(CuFe) <sub>12</sub> As <sub>4</sub> S <sub>13</sub>	Cu-Ag e.g. Equity Silver mine, Canada	-

### 3. Health effects of arsenic

#### 3.1 Toxicity

Arsenic has historically been used as a poison because of its toxicity and lack of taste and odour (Ferguson and Gavis, 1972; Whitacre and Pearse, 1974; Korte and Fernando, 1991). The degree of toxicity of arsenic compounds is inversely proportional to the rate of excretion from the body. The speciation of arsenic determines its toxicity in the environment. All inorganic forms of arsenic are more toxic than organic forms of arsenic to a degree related to their solubility (Hart, 1982). As(III) is reportedly 25-60 times more toxic than As(V) and several hundred times more toxic than organic arsenic compounds (Ferguson and Gavis, 1972; Morrison, et al., 1989). A dose of 100 mg of arsenic trioxide, As<sub>2</sub>O<sub>3</sub> (as As(III)), induces toxic effects in humans, while 130 mg is fatal (Hart, 1982).

A substance is defined as toxic if it injures the growth or metabolism of an organism when supplied above a certain concentration. One of the most common modes for an element to induce toxicity is to inactivate enzyme systems (Whitacre and Pearse, 1974). Enzymes are proteins, or amino acids, which act as biological catalysts and accelerate chemical reactions in organisms. The increased toxicity of As(III) relative to As(V) is due to the former's ability to react with sulfhydryl groups of the amino acid cysteine to block the enzyme's activity. This increases the residence time of As(III) in the body (Ferguson and Gavis, 1972; Korte and Fernando, 1991). There is some evidence that As(V) is an essential trace element in human and some animal physiologies (Hopkin, 1988). However, evidence also indicates that As(V) substitutes for phosphate in living organisms thereby preventing the functioning of phosphates

responsible for the metabolism of foodstuffs (Whitacre and Pearce, 1974; GESAMP, 1988).

Arsine gas,  $\text{AsH}_3$  (as oxidation state -III) is the most dangerous arsenic compound. Its toxicity is due to its ability to break down red blood cells in the human body (Whitacre and Pearce, 1974). However, arsine gas rapidly decomposes in the presence of light and moisture and can be effectively dissipated in the environment (Fowler, 1976).

Arsenic is also toxic to animals with a lethal dose of  $\text{As}_2\text{O}_3$  reported for most laboratory animals between 11 and 150 ppm. A toxic dose of  $\text{As}_2\text{O}_3$  for sheep and horses is approximately 10 to 15g and for cows, approximately 15 to 30g. However, arsenic concentrations in the range of 10 to 20 ppb appear to be well-tolerated by animals (Department of Manufacturing and Industry Development, 1991). The recommended maximum arsenic concentration for irrigation waters on farmlands is 0.1 mg/L As (Table 3).

A considerable amount of literature exists for the effects of arsenic exposure to aquatic animals and plants (Woolson, 1974; GESAMP, 1988; Cullen and Reimer, 1991). Arsenic is accumulated in aquatic organisms to a greater extent than terrestrial organisms, with the greatest accumulation occurring in plants at the lowest end of the food chain (Woolson, 1974; Andreae and Klumpp, 1979). Invertebrates are generally more sensitive to arsenic than adult fish and marine organisms can accumulate more arsenic than their freshwater counterparts. However, although concentration of arsenic appears to occur by some aquatic organisms, arsenic is not usually bio-magnified, with arsenic levels in organisms rarely exceeding 160 ppm. For example, arsenic concentrations between 3 to 49 ppm have been measured in shrimp. At these levels, such organisms are not toxic to humans and the arsenic is readily excreted (Woolson, 1974). In plants, however, higher concentrations of arsenic have been measured, for example 11 to 1450 ppm As recorded in some lakeweeds and 60 to 142 ppm As found in seaweed (Woolson, 1974). The U.S. EPA has estimated that acute toxicity to freshwater invertebrates commences at concentrations of approximately 0.8 mg/L (Department of Manufacturing and Industry Development, 1991). The recommended maximum concentration for arsenic in fresh and marine water is 0.05 mg/L As (Table 3).

Arsenic trioxide has been used as a herbicide to sterilise soils. The toxicity of arsenic to plants is a function of the species of arsenic present and the amount of water soluble arsenic which is available for plant uptake. Concentrations of 3 to 10 ppm of soluble arsenic have been recorded as toxic to crops of barley (Department of Manufacturing and Industry Development, 1991).

### **(i) Human toxicity**

Exposure to arsenic is unavoidable, as arsenic is ubiquitous and naturally occurring in the environment. The toxicity of arsenic for humans depends on its chemical form, the route of ingestion, dose, exposure, age and sex of the person (Hart, 1982). The main routes for the transfer of arsenic from the environment to humans occur via inhalation and ingestion. It is estimated that a person consumes 45 µg/day of arsenic in food and 10 µg/day from drinking waters. Inhalation pathways occur mainly in areas of potential occupational exposure to arsenic, such as smelters and arsenic powder producing plants. Ingestion of elevated concentrations of arsenic occurs primarily through the consumption of arsenic contaminated drinking water (Ferguson and Gavis, 1972; Korte and Fernando, 1991). Other forms of ingestion include incidents of arsenic accumulation in shellfish (up to 100 ppm (Whitacre and Pearse, 1974)) and in wines (0.53 mg/L), where arsenical pesticides (such as lead arsenate) have been used (Woolson, 1983). These levels of contamination compare to the recommended maximum levels for arsenic in drinking waters of 0.05 mg/L As (Table 3).

Arsenic is usually rapidly cleared from the body through excretion. However small amounts of arsenic are retained for longer periods in selected tissues such as hair, fingernails, muscle, liver and kidney (Department of Manufacturing and Industry Development, 1991).

Arsenic toxicity manifests itself as cutaneous deposits in the skin and other keratinous tissues, for example finger nails and hair. Severe cases of arsenic poisoning produce pigment disorders, skin lesions, gangrene (Blackfoot disease) and ultimately skin cancer (Utidjian, 1976, Tseng, 1977). Arsenic also affects the central and peripheral nervous system, it is cardiotoxic and is toxic to the kidneys (Utidjian, 1976).

### **3.2 Carcinogenesis**

Recent literature has confirmed that certain species of arsenic can be carcinogenic to humans (Buat-Menard et al., 1987; Stöhrer, 1991). As(III) is generally accepted to cause skin cancer and As(V) has been linked to lung cancer in exposed occupational groups, for example copper smelter (Perschagen et al., 1977) and pesticide workers (Stöhrer, 1991). These findings present serious implications for the mining industry in terms of increasing the regulation of arsenic emissions. These regulatory issues are discussed further in Section 9.

Considerable confusion has been associated with research concerning the carcinogenic effects of arsenic (Perschagen et al., 1977; Korte and Fernando, 1991). This has been mainly due to the inability to reproduce point mutations in animal experiments, which are seen as the most sensitive indicators of gene damage leading to cancer. Alternative experiments have produced indications of other effects of genetic damage by arsenic exposure (Stöhrer, 1991). This apparent contradiction has

been explained recently by the understanding that arsenic is an *indirect* carcinogen, inducing damage in genes controlling proliferation, differentiation, and recombination (Stöhrer, 1991).

The number of documented incidents of cancer as a result of exposure to arsenic is few. Less than one thousand cases of arsenical cancer associated with environmental or occupational exposure to arsenic have been described in the medical literature over the past one hundred years (Stöhrer, 1991). Two groups of arsenical cancer have been documented in the U.S. One group of 50 people with lung cancer is linked to past occupational exposure from lead smelter and pesticide industries. A second group is related to populations exposed to elevated arsenic in drinking water (above 0.05 mg/L) and indicates possibly ten incidents of arsenical disease (Stöhrer, 1991).

In a study of arsenic concentrations in wells used for drinking water supply in Nova Scotia, Canada, 66 out of 94 samples (i.e. 70%), exceeded Canadian maximum acceptable drinking water guidelines of 0.05 mg/L (Meranger et al., 1984). It was reported that 8 out of 42 people drinking groundwater in Nova Scotia with arsenic concentrations of 0.05 to 0.1 mg/L had mild clinical symptoms possibly related to arsenic poisoning (Korte and Fernando, 1991). Other studies have reported ingestion of drinking water with arsenic concentrations of 0.05-0.25 mg/L with no adverse health effects (Ferguson and Gavis, 1972).

### 3.3 Dose-response data and guidelines for arsenic

The best dose-response database for arsenical diseases, including Blackfoot disease and cancer, is Taiwan. Over a period of 45 years, approximately 40,000 people drank groundwater containing arsenic at concentrations of 0.01 to 1.82 ppm. More than 7,000 people developed arsenical diseases (Tseng, 1977). A second study in Bengal documented arsenical disease in a group of individuals consuming groundwater with arsenic concentrations of approximately 200 µg/L while a control group consuming 31-37 µg/L As in groundwater exhibited no adverse health effects. Figure 1 shows the dose-response data from these two studies and indicates that the lowest estimate of the threshold for adverse health effects due to exposure to arsenic is 100 µg/L (0.1 mg/L) (Stöhrer, 1991). At a consumption of approximately 4 litres of water per day, more than 400 µg per day (0.4 mg) is required to cause arsenical disease in humans (Stöhrer, 1991). The threshold dose of 100 µg/L compares to a much lower regulated concentration of 50 µg/L of arsenic in drinking water (0.05 mg/L) (Table 3).

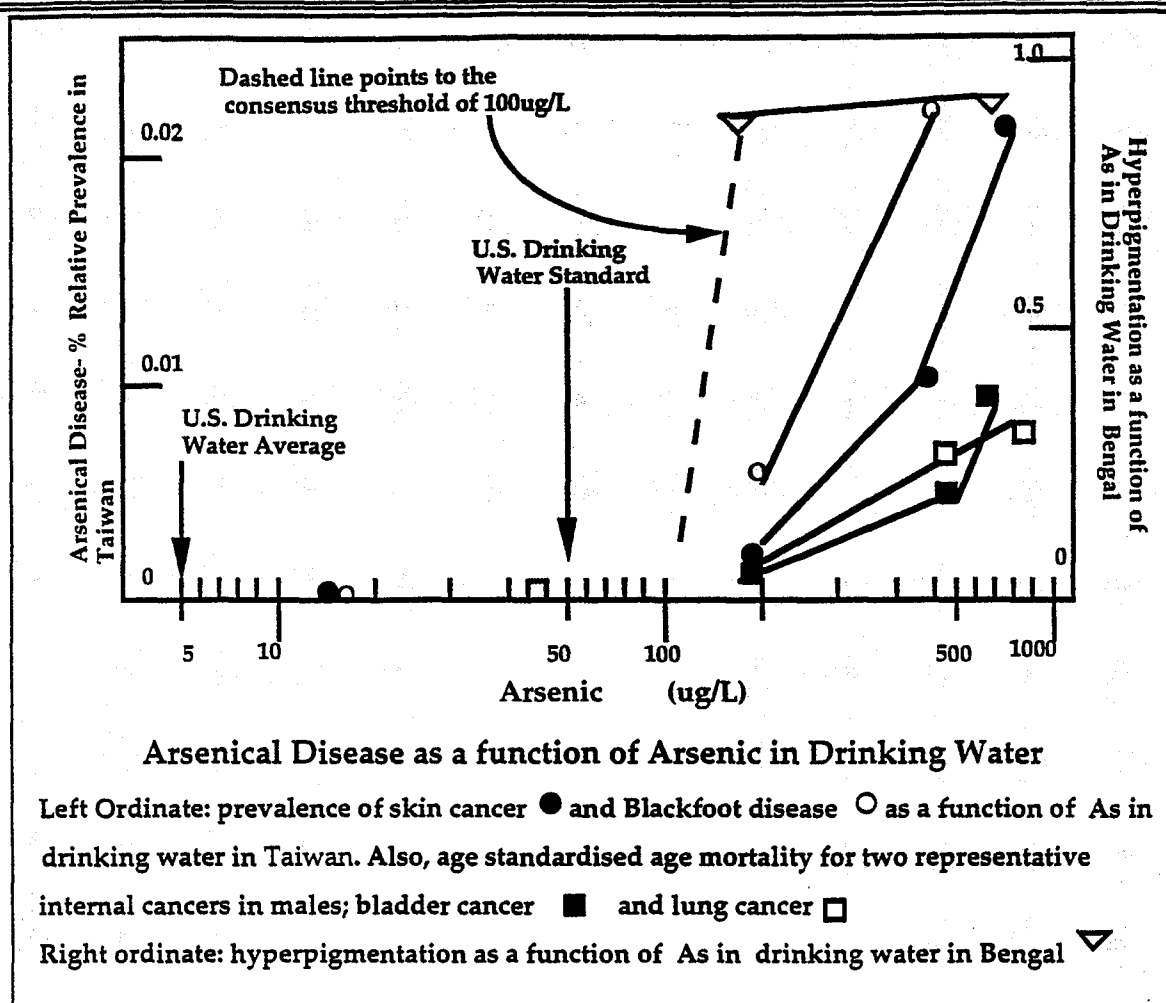


Figure 1: Dose-Response data for Taiwan and Bengal for arsenic disease and cancer (Stöhrer, 1991)

Stöhrer (1991) argues that arsenic may provide a very useful example for determining safe levels for indirect carcinogens in the environment, as gene induction provides a molecular basis which can be studied in dose response relationships. The level of arsenic exposure is related to the onset of skin pigmentation which can be measured and used as an indication of threshold limits for arsenic prior to the onset of skin cancer. This is a unique and measurable characteristic for a carcinogen to display. However, current regulations do not accept safe levels of carcinogens in the environment (Stöhrer, 1991). As a result, many governments are in the process of reviewing the safety limit for arsenic (Pearce, 1993) and it is expected that future maximum acceptable concentrations for arsenic in the environment will be decreased (see Section 9).

*Arsenic in the Gold and Base-Metal Mining Industry*

*Table 3: Recommended arsenic concentration guidelines for drinking water, soil landfill and the aquatic environment (Environmental Geochemistry International*

*(EGI), 1994).*

Arsenic Concentration Guideline		WHO (1992)	ANZECC (1992)	NSW SPCC (1990)	Vic EPA (1991)	U.S. EPA (1986)	Dutch (1992)
Media							
Water (mg/L)	Drinking	0.05	0.05	0.05		0.05	
	Stock watering Irrigation Ground Water		0.2 0.1	0.2 0.1			0.1
Soil Quality (mg/kg dry weight)	Environmental Investigation Criterion		20				30
	Human Health Investigation Criterion		100				
Landfill (mg/kg dry weight)	Clean Fill				30		
	Acceptable Low Level Contaminated Soil				300		
Aquatic Environ. (mg/L)	Fresh Water-Acute		0.05	*0.36 #0.19		*0.36 #0.19	
	-Chronic Marine Water-Acute -Chronic		0.05	*0.069 #0.036		*0.069 #0.036	

\* The 1-hour average concentration of arsenic should not exceed this value more than once every 3 years.

# The 4-day average concentration of arsenic should not exceed this value more than once every 3 years.

WHO-World Health Organisation

ANZECC- Australian and New Zealand Environmental Conservation Council

NSW SPCC-NSW State Pollution Control Commission

Vic EPA-Victoria Environmental Protection Authority

US EPA- United States Environmental Protection Agency

Dutch B-Dutch Guidelines for the recommendation of further investigation

#### 4. Environmental geochemistry in soils and waters

The biogeochemical cycling of arsenic in soils and waters is complex. Arsenic displays a wide range of reactivity in the environment, participating in oxidation-reduction, acid-base, ligand exchange, precipitation, adsorption and methylation-demethylation reactions (Ferguson and Gavis, 1972; Kersten, 1988). Figure 2 shows the primary reactions involving arsenic in soils and waters. A brief introduction to arsenic geochemistry is presented in this section.

Arsenic exists in natural waters, soils and sediments in four principal oxidation states, -III, 0, III, V, as arsine gas ( $\text{AsH}_3$ ), sulphides and oxides, arsenite ( $\text{AsO}_3^{3-}$ ) and arsenate ( $\text{AsO}_4^{3-}$ ), respectively. Arsenic occurs in the environment mainly as the inorganic arsenic oxides, arsenite and arsenate, and its simpler methylated forms shown in Figure 2 (Ferguson and Gavis, 1972). Each species of arsenic displays a unique environmental chemistry with different degrees of mobility, bioavailability and toxicity (Boyle and Jonasson, 1973; Hart, 1982; Adriano, 1986). Arsenic speciation is in turn controlled by environmental parameters such as Eh, pH, iron oxide, clays, soil texture and organic matter (Abdelghani et al., 1981; Cherry et al., 1986c; Brannon and Patrick, 1987; Korte and Fernando, 1991).

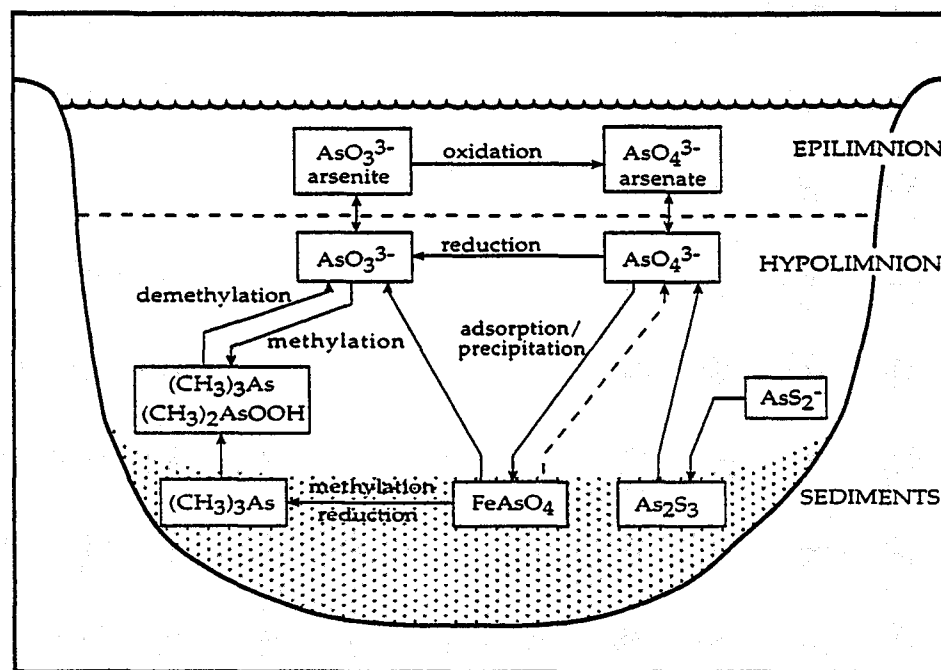


Figure 2: The geochemical cycle of arsenic in natural waters (Ferguson and Gavis, 1972).

In natural waters, arsenic is mainly found at low concentrations (averaging 1.7 ppb) as inorganic As(III) and As(V) acids (Ferguson and Gavis, 1972). In oxygenated environments, As (V) (as  $\text{AsO}_4^{3-}$ ) is the stable species and is shown in the Eh-pH diagram in Figure 3. In anaerobic environments, such as flooded soils and groundwater below the base of oxidation, As (III) (as  $\text{AsO}_3^{3-}$ ) is the dominant form

(Masschelyn et al., 1991). Arsenic is generally more mobile in reduced than oxidised environments (Hermann and Neumann-Mahlkau, 1985). As(III) is more toxic and has been found to be 4-10 times more soluble in soils than oxygenated As(V) species (Adriano, 1986). At very low Eh values, arsine gas ( $\text{AsH}_3$ ) and arsenic sulphides such as arsenopyrite ( $\text{FeAsS}$ ) may be formed (Adriano, 1986). The rate of oxidation of As(III) to As(V) is relatively high (hours) and increases with pH, salinity and methylation reactions of arsenic by a number of soil organisms in aerobic and anaerobic environments (Woolson, 1983).

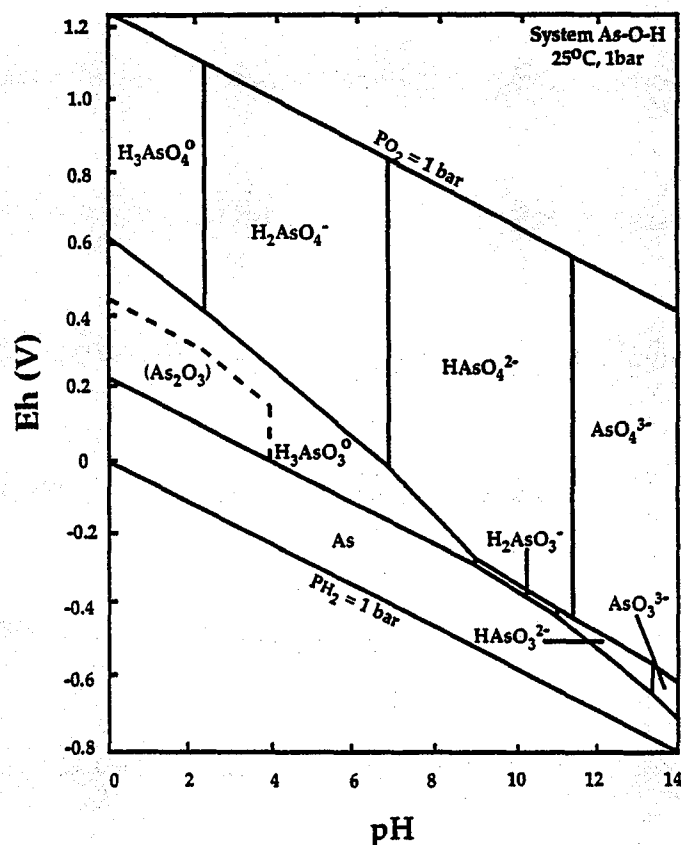


Figure 3: Eh-pH diagram for the system As-O-H at 25°C and 1 atmosphere. The oxidation state of arsenic species is tabulated below (Ferguson and Gavis, 1972; Bowell et al., 1994).

As(0)	As(III)	As(V)
As(c)	$\text{As}_2\text{O}_3(\text{c})$	$\text{H}_3\text{AsO}_4^0$
	$\text{H}_3\text{AsO}_3^0$	$\text{H}_2\text{AsO}_4^-$
	$\text{H}_2\text{AsO}_3^-$	$\text{HAsO}_4^{2-}$
	$\text{HAsO}_3^{2-}$	$\text{AsO}_4^{3-}$
	$\text{AsO}_3^{3-}$	

A variety of organisms, such as bacteria and fungi, have the ability to oxidise/reduce and methylate/demethylate arsenic species in both aerobic and anaerobic environments (Piver, 1983; Masscheleyn et al., 1991). In low Eh

environments, such as flooded soils, methanogenic bacteria can reduce and/or demethylate As(V) to As(III) and methylarsenic acid(III) or dimethylarsenic (cacodylic) acid(I) (Korte and Fernando, 1991). In oxidised environments, bacteria and fungi have been described which oxidise and methylate As(III) to As(V) (Cullen and Reimer, 1991). In addition, the microbial reduction of methylarsenic acid(III) or cacodylic acid to volatile alkylarsines and the subsequent emission of these compounds to the atmosphere is frequently observed (Adriano, 1986; Korte and Fernando, 1991; EPA, 1992).

The chemical behaviour of arsenicals (arsenate, arsenite and methyl-arsenic compounds) is complex in soil ecosystems. These compounds can react strongly with clay minerals, iron and aluminium hydroxides, organics, carbonates and sulphides, to form very insoluble complexes, such as ferric arsenate (scorodite,  $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$ ) and arsenic-bearing oxy-hydroxides (Piver, 1983). Different clay types adsorb arsenic to varying degrees. For example, compared to kaolinite, montmorillonite adsorbs more As(V) than As(III) (Adriano, 1986). Soil texture also affects arsenic mobility (Adriano, 1986). As(III) has been found to be 5-8 times more mobile than As(V) in sandy soils (Piver, 1983; Woolson, 1983). The amount of total arsenic present in solution is greater in sandy soils with low clay and organic contents.

To a large degree, soil Eh determines the ratio of As(III):As(V). However, Eh is a function of a range of parametres including iron content, pH, microbial population and moisture content (EPA, 1992). Generally, as Eh decreases below 300mV, due to flooding or changing conditions, the arsenite content of soils (and waters) increases and the arsenate content decreases.

In aerobic soils, the adsorption reactions described above provide a major sink for arsenic, particularly via adsorption onto iron oxides (Braman, 1983; Brannon and Patrick, 1987). However, fluctuations in oxidation levels and changes in pH can remobilise and release arsenic into the environment (Abdelghani et al., 1981). The increased mobility of arsenic in sands compared to clays suggest the potential risk for arsenic contamination in sandy soils should be considered in areas of arsenic disposal. The ultimate sink for arsenic is adsorption onto ocean sediments. These are important considerations for the development of safe methods for immobilisation and disposal of arsenic in the environment and will be discussed further in Section 8.

## **5. Natural emissions**

Arsenic has both natural and anthropogenic (synthetic) sources in the environment and can enter in gaseous, dissolved and solid states (Woolson, 1983). The natural sources of arsenic in the environment are described in this section. Anthropogenic sources of arsenic, including arsenic emissions from the gold and base-metal mining industry, are discussed in the following section. There is some doubt concerning the relative contribution of natural and anthropogenic emissions of

arsenic, due mainly to the difficulty in estimating the quantity of natural releases, however, Buat-Menard et al. (1987) estimate a ratio of 60:40, respectively (GESAMP, 1988).

Arsenic is ubiquitous in the earth's crust, with average concentrations in soils and sediments of 1.5-2 ppm (Woolson, 1983; GESAMP, 1988) and in ocean waters of approximately 2 ppb (Ferguson and Gavis, 1972; Table 1). Arsenic is frequently associated with sulphur where it can form arseniferous sediments (100-300 ppm) or mineralised zones with higher arsenic concentrations (up to 4 % in copper-arsenopyrite ores) (Table 2). Natural emissions to the atmosphere account for approximately 60% of arsenic emissions and include volcanoes, erosion of sediments and low temperature volatilisation of organic material (Table 4). Ferguson and Gavis (1972) have estimated that the erosion of arseniferous sediments contributes a total of 41,000 tonnes of arsenic per annum to rivers and oceans, representing approximately 25% of total arsenic emissions to land and waters (Table 5).

Contamination of groundwater for drinking supplies occurs in geological environments with naturally high concentrations of arsenic. As this may pose a significant problem for human health, water used for drinking in these areas requires careful monitoring of arsenic concentrations or remediation to remove arsenic (Piver, 1983).

*Table 4: Natural and anthropogenic arsenic emissions to the atmosphere (Chilvers and Peterson, 1987)*

<b><i>ATMOSPHERIC EMISSIONS</i></b>		
<b><i>Natural Emissions</i></b>	<b><i>Tonnes As per year</i></b>	<b><i>%</i></b>
<i>Low-temp volatilisation of organic matter</i>	26,200	58
<i>Volcanic</i>	17,150	38
<i>Erosion of sediments</i>	2,132	4
<b><i>TOTAL</i></b>	<b><i>45,482</i></b>	<b><i>100</i></b>

<b><i>Anthropogenic Emissions</i></b>	<b><i>Tonnes As per year</i></b>	<b><i>%</i></b>
<i>Copper smelting</i>	12,080	43
<i>Non-ferrous processing</i>	2,270	8
<i>Coal combustion</i>	6,240	22
<i>Pesticides, glass</i>	7,480	27
<b><i>TOTAL</i></b>	<b><i>28,070</i></b>	<b><i>100</i></b>

Table 5: Natural and anthropogenic arsenic emissions to land and water (Ferguson and Gavis, 1972; Chilvers and Peterson, 1987)

<b>LAND AND WATER EMISSIONS</b>		
<b>Natural Emissions</b>	<b>Tonnes As per year</b>	<b>%</b>
<i>Erosion of sediments</i>	41,000	100

<b>Anthropogenic Emissions</b>	<b>Tonnes As per year</b>	<b>%</b>
<i>Copper smelting</i>	86,275	56
<i>Non-ferrous processing, smelting</i>	15,480	10
<i>Coal Combustion</i>	35,100	23
<i>Herbicides, glass manufacturing, etc</i>	16,560	11
<b>TOTAL</b>	<b>153,415</b>	<b>100</b>

## 6. Anthropogenic sources

The estimated annual emissions of arsenic from the primary anthropogenic sources are presented in Tables 4 and 5. At present, anthropogenic sources contribute about 40% of arsenic emitted to the atmosphere and 75% of arsenic entering the world's oceans. The majority is derived from smelting operations, mining processes and pesticides. Copper smelting accounts for over 40% of anthropogenic arsenic emissions to the atmosphere and nearly 60% of arsenic emissions to waters. The production of non-ferrous metals other than copper (for example gold and nickel), constitutes approximately 10% of anthropogenic emissions to both air and water. Emissions from coal combustion account for over 20% of arsenic emissions to both air and water.

The remainder of the anthropogenic emissions are from industrial sources, such as pesticides, sewage effluents and sludges and urban stormwater (Hart, 1982). Arsenic is used in the manufacture of arsenical pesticides and wood preservation, of which approximately 10% may be volatilised to air by anaerobic and aerobic bacterial activity. Arsenic is also used as a dessicant in cotton production and in glass manufacturing (Pacyna, 1987).

Geothermal production of energy also has the capacity to release significant amounts of arsenic to surface water. Wells used for geothermal energy production in New Zealand tap arseniferous sediments at depth and can discharge from 2 to 20,000 ppm (2%) arsenic in some areas (Aggett and Aspell, 1980; Piver, 1983).

### 6.1 Arsenic emissions from the non-ferrous metals industry

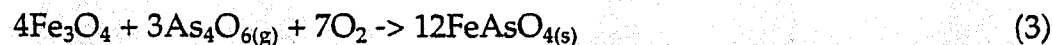
In the non-ferrous metals industry, arsenic is emitted to the environment from smelting, refining and processing of ore (Table 5) through processes including;

- volatilisation during smelting and refining,
- disposal of flue dusts and solid residues from roasting and smelting,
- acid mine drainage (AMD) from tailings dams and stockpiles,
- mine water seepage from mines and waste water processing,
- contamination of surface waters from mine and smelter discharge.

Large stockpiles of waste arsenic from the mining and processing of ores currently lie untreated in many countries around the world, including Australia (Piret and Melin, 1989). These stockpiles exist because the world demand for  $\text{As}_2\text{O}_3$  is smaller than the total amount of arsenic produced. Arsenic is a relatively low value by-product from metal ore processing and therefore, the recovery of arsenic from stockpiles is not usually economic (EPA, 1992).

For example, at the Windarra Gold operation in Western Australia, refractory gold ore has been mined with grades of 6-8 g/t Au and 0.1-0.5% As. Approximately 40-50% of the gold occurs as sub-micron phases within arsenopyrite and requires roasting and leaching to liberate the gold (Quadrio and Kellett, 1984). The roasting process at Windarra is shown in Equations 1 to 3 and produces  $\text{As}_2\text{O}_3$  and ferric arsenate ( $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$ ) as a by-product of the gold extraction. The Windarra Gold operation is currently being decommissioned, and all arsenic at this site has been sold within Australia as product.

*Roasting of arsenopyrite at approx. 700°C : (Quadrio and Kellett, 1984 )*



In addition, strict environmental constraints on arsenic emissions and disposal have contributed to the creation of large arsenic stockpiles (EPA, 1992). Moreover, increasingly stringent environmental regulations for the transport of toxic wastes (as a result of the Basel Convention, see section 9 and Appendix 1), will prohibit the international trade in such wastes. In the future, management of arsenic wastes will require immobilisation and disposal of the waste, or the development of new markets for the waste within the country of origin.

As well as the stockpile issue, arsenic emissions from smelting and from leachate generated from acid mine drainage of arsenic-bearing sulphides, are two additional areas of concern to industry, government regulating authorities and the general

public. These areas are discussed below to provide an indication of the level of arsenic contamination on mining sites.

### **(i) Smelting**

Arsenic pollution of the atmosphere results mainly from the gaseous emissions of non-ferrous metal smelters and refineries, coal combustion and pesticide use (Table 4). During the smelting process, arsenic contained as a trace metal in a variety of ores, in particular copper, gold and nickel, is volatilised and driven off into the flue gases. The arsenic concentration of smelter dusts may range from a few percent to 50% (Piver, 1983). These dusts are re-roasted and arsenic is fractionally sublimed and cooled in chambers called "kitchens". Without adequate filtering these dusts can escape into the atmosphere (Nelson, 1976). The solid residue in the kitchens is scraped away and sold as  $As_2O_3$  for industrial or pesticide manufacturing purposes. There is very little information on chemical speciation of arsenic in the atmosphere, however it appears that trivalent, inorganic arsenic is the dominant form near emission sources such as smelters (Pacyna, 1987).

The behaviour of arsenic in the atmosphere depends on the physical and chemical properties of the arsenic particulates and on local meteorological conditions. Arsenic, and other volatiles, are condensed on small particles which are coated with  $H_2SO_4$ . Temperature inversions or periods of stagnant weather will tend to concentrate emissions close to the source, while periods of windy weather will disperse arsenic emissions over great distances (Piver, 1983). Particulate matter has an estimated residence time in the atmosphere of 7 days (Pacyna, 1987), during which time arsenic can be dispersed over thousands of kilometers. In a study of contaminated soils in Norway, Steinnes (1987) determined arsenic (and other heavy metals) to have travelled to Norway via atmospheric emissions from central Europe.

Arsenic-bearing particles are removed from the atmosphere by precipitation or by gravity settling, eventually contaminating surrounding land and water. Generally, arsenic emissions from point sources such as smelters, reach a maximum concentration around the source and gradually decrease in concentration with distance from the source. At Puget Sound, U.S.A., sediments contain up to 10,000 ppm arsenic, probably derived from emissions from a nearby smelter (Piver, 1983).

### **(ii) Acid mine drainage (AMD)**

Arsenic released into the environment as a result of Acid Mine Drainage (AMD) from tailings and waste rock piles is a significant problem (Piver, 1983), particularly in areas of gold mining where gold is associated with arseniferous ores. For example in the U.S.A., approximately 7,000km of streams are currently affected by AMD (Ferguson and Erikson, 1988). Over a period of 100 years, Whitewood Creek, South Dakota, U.S.A., received approximately 1 billion tonnes of arsenic-rich tailings from gold

mining activities, a portion of which has been deposited on the floodplain of the creek (Cherry et al., 1986a,b,c). In Nevada, Woolson (1983) reported that leaching of acidic fluids from tailings dams contaminated surrounding soils with concentrations of between 170 and 6000 ppm As in areas close to the dam. In Canada, 315 million tonnes of waste rock (or approximately an area of 15,000 ha) have been produced at 56 base-metal mines. Of these mines, 75% produce AMD and the amount of AMD producing waste rock is increasing by 25 million tonnes per year (Siwik et al., 1989; Peeling et al., 1992). In Australia, major AMD pollution has occurred at base-metal mines in Mt Lyell, Tasmania and in Mt Morgan, Queensland. Only recently, in 1995, has preliminary research been undertaken at Mt Lyell to identify the extent of the AMD pollution (pers. comm. Barry Carbon, Exec. Director, Supervising Scientist, EPA, 1995).

Pollution from tailings and waste dumps will continue well into the future as arsenic-bearing tailings progressively oxidise and release arsenic into the environment (Brooks et al., 1982). Tables 6 and 7 present a number of examples of arsenic concentrations in tails and mine groundwater in Australia and overseas. The arsenic concentrations at these mine sites are significant when compared to both the maximum legislated concentration in drinking water of 0.05 mg/L As (Table 3) and to the maximum concentration of 30 ppm As in sediments used for clean fill. The information presented in Table 6 implies that a large proportion of houses in areas of past gold mining, for example Bendigo, Victoria, are built on fill from tailings containing up to 100 times the legislated arsenic limit for clean fill.

*Table 6: Arsenic contents of tailings in Victoria, New Guinea, Canada and the U.S.A. (Cherry et al., 1986; Department of Manufacturing and Industry Development, 1991)*

<u><i>Arsenic Concentrations in Tailings Dams</i></u>	<u><i>As (ppm)</i></u>
<i>Wattle Gully (in &lt;63<math>\mu</math> fraction)</i>	<i>28-10192 (av 3761)</i>
<i>Bendigo</i>	<i>similar range</i>
<i>Bendigo residential area with mine waste as fill</i>	<i>up to 3100</i>
<i>Stawell Gold Mine</i>	<i>1160 to 2650</i>
<i>Swifts creek</i>	<i>300 to 15000</i>
<i>Nova Scotia, Canada</i>	<i>3750 to 1500</i>
<i>Papua New Guinea</i>	<i>400 to 600</i>
<i>Western Australia</i>	<i>3400</i>
<i>Whitewood Creek, Dakota, U.S.A.</i>	<i>500 to 5000</i>

**Table 7: Arsenic contents in groundwater from abandoned mines in Victoria (Department of Water Resources Victoria, undated)**

<u><i>Arsenic Levels in Groundwater from Abandoned Mines in Victoria</i></u>	<u><i>As (mg/L)</i></u>
<i>Woods Point Morning Star mine</i>	300
<i>St Arnaud, Burkes Flat Mine</i>	0.1 to 3.5
<i>Bendigo, Central Deborah Mine</i>	1.8
<i>Sandy Creek, BF Mine</i>	0.78
<i>Bendigo, Shamrock workings discharge to Bendigo Ck</i>	0.5
<i>Brunswick Mine</i>	0.38
<i>Ballarat, Sovereign Hill, bore hole at 700m</i>	5.0

## 7. Recovery of Arsenic

Arsenic is a low value by-product of a variety of gold and base-metal ores, including gold associated with arsenopyrite, arsenic-bearing nickel ores and copper sulphide ores (EPA, 1992). In many cases, the arsenic contains impurities such as iron, selenium, bismuth, lead, antimony. These elements can often be uneconomic to remove and the arsenic is stockpiled as waste. However, where arsenic of a sufficient purity is produced (purity greater than approximately 95%) the arsenic may be economically recovered for use, primarily in the manufacture of the arsenical wood preservatives, chromated copper arsenate (CCA) and ammoniacal copper-zinc arsenate (ACZA) (EPA, 1992). The source for the arsenic may be smelter flue dust, slimes, sludges and filter cakes from metal refineries, arsenopyrite wastes or concentrate from mines, in particular from the processing of copper sulphide ores and other non-ferrous metals (gold, nickel, zinc).

The major source countries for arsenic as a commercial product in 1991 are shown in Table 8. Prior to 1991, the U.S. produced a significant amount of  $As_2O_3$  from the ASARCO Incorporated smelter in Tacoma, Washington (7,800 tonnes in 1981). The  $As_2O_3$  was used primarily for the production of agricultural pesticides. In 1985, the smelter was shut down because of the need to comply with regulations for atmospheric emissions of arsenic and sulphur dioxide, and operations were moved to the Philippines (EPA, 1992; Tran, 1993). Until 1993, Australia was a minor producer of  $As_2O_3$  from the Windarra Gold Operation, in Western Australia (pers. comm. W. Cunningham, Western Mining Corporation Limited, 1993).

Table 8: Commercial producers of arsenic trioxide (EPA, 1992)

ARSENIC PRODUCERS	
Country	1991 Tonnage
China	10,000
Chile	7,000
U.S.S.R	7,000
Philippines	5,000
Mexico	4,960
France	3,000
Belgium	3,000
Sweden	2,500
U.S.	(ceased production in 1985)
<b>TOTAL</b>	<b>42,460</b>

### 8. Immobilisation and treatment technologies

As indicated, the supply of arsenic is much greater than the world demand and as a result, large amounts of waste arsenic have been stockpiled. Arsenic is relatively soluble in water and solubility increases with temperature at both pH extremes (Cherry et al., 1986c). Exposure to water can mobilise arsenic into solution with the consequent potential for groundwater contamination. Therefore, arsenic waste cannot be disposed of in areas where it is likely to come into contact with water. Arsenic waste thus requires immobilisation or treatment to remove arsenic from the waste stream. This section describes the current methods available for the treatment of arsenic waste for subsequent immobilisation and disposal in an environmentally responsible manner.

Research into immobilisation and treatment of arsenic waste is currently being conducted at the Australian Nuclear Science and Technology Organisation (ANSTO) in a number of areas including biological remediation, cement and polymer encapsulation of arsenic-bearing wastes and iron adsorption technology (Tran, 1993; Zuk, 1993). Biological remediation, which is still far from being technologically feasible, uses algae and bacteria to extract dissolved arsenic. Cement encapsulation successfully immobilises arsenic wastes and tentative costings at \$400/tonne indicate that this technology may be practical for disposal of small volumes of arsenic waste (pers. comm. Dr Ging Khoe, ANSTO, 1993). Iron arsenate adsorption technology requires the oxidation of As(III) to As(V) prior to the adsorption process and appears to be the most promising technology in terms of long-term stability of mine wastes and the removal of arsenic from contaminated waters (pers. comm. Dr Ging Khoe, ANSTO, 1993). This process will be reviewed in the next section.

### 8.1 Iron arsenate adsorption technology

The precipitation of relatively insoluble calcium arsenate with lime ( $\text{CaCO}_3$ ) has been widely practiced as a method for the disposal of arsenic-bearing wastes in the mining industry (Laguitton, 1976). However, in Australia, Robins (1980) and Robins and Tozawa (1982) have shown that calcium arsenate precipitates gradually decompose in the presence of dissolved carbon dioxide in groundwater to release arsenic into the surrounding environment. These findings present a large problem for the mining industry in terms of ameliorating existing calcium arsenate disposal sites. For example, several tailings dams in the U.S. release significant concentrations of arsenic into groundwater (EPA, 1992). As a result, the mining industry is faced with developing new methods for the stabilisation and disposal of arsenic waste.

The adsorption of arsenic on ferric oxy-hydroxide at pH 4, is considered to be the most effective material for binding As(V) and to a lesser extent As(III) (Robins et al., 1988; EPA 1992). Work undertaken at INCO (Harris and Krause, 1993) and at Noranda (Harris and Monette, 1989), demonstrated improved stability of iron arsenates with higher Fe:As molar ratios. This study found that Fe:As molar ratios of at least 4 were required in solids in order to maintain arsenic concentrations in leachate below 5 mg/L at pH's of 3 to approximately 7. In addition, small amounts of Cu, Cd and Zn added to solution during the adsorption process were found to increase adsorption of arsenic over a pH range of 4 to 10 (Harris and Monette, 1989). Moreover, Emmett and Khoe (1994) found the addition of non-toxic cations, such as Ca and Mg, produced a similar increase in the stability of the arsenic-bearing ferric oxy-hydroxide.

In work conducted on flue dust samples from roasters at the Windarra Operation, Western Australia, Kyle and Lunt (1991) showed the formation of ferric arsenate precipitate produced products with very low solubilities, ranging from 2 to 4 mg As/L in samples containing Fe:As molar ratios of 2:1 and 4:1, respectively. However, it was concluded to be an uneconomic method for immobilisation of arsenic at this site because of the large amounts of lime and ferric sulphate required for the stabilisation process, shown in Equation 5. For example, only 200g of arsenic trioxide were precipitated with 1500g of ferric sulphate and 910g of lime. However, the possibility of using the iron arsenate technology in areas where a large iron component naturally exists in the ore was not discussed by Kyle and Lunt (1991). At sites with high iron contents in the ore, the ferric arsenate technology may prove to be efficient for arsenic removal and treatment. Moreover, the expected increase in regulation of hazardous wastes and the control of emissions may force industry to adopt more expensive technologies for immobilisation of such wastes.

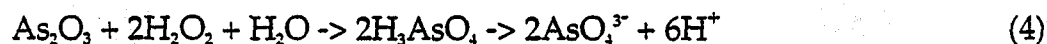
Research by Khoe et al. (1991) found no complexation of ferrous iron with arsenate in contrast to significant complexation of ferric iron with arsenate. These results are important considering that a significant proportion of total iron in process

liquors from mine wastes is in the divalent state and would therefore require oxidation of the iron prior to the precipitation process (Khoe et al., 1991).

The adsorption of arsenic on iron oxy-hydroxides is most efficient when arsenic is present as As(V), i.e. as the anion complex  $\text{AsO}_4^{3-}$ . Tran (1991) recommended the oxidation of As(III) to As(V) using hydrogen peroxide (or sodium hypochlorite) and Fe:As molar ratios greater than 5 (shown in Equation 4) prior to the precipitation of As(V) with ferric sulphate (shown in Equation 5). Ongoing work at ANSTO is researching the effectiveness of photolytic oxidation of As(III) to As(V) (pers. comm. Dr Ging Khoe, 1994). This process essentially requires the exposure of the sample to sunlight and has been found to be very effective in oxidising arsenic in mine waste waters (Ringwood, 1995). In addition, this process would remove the need for chemical oxidants for the adsorption process, thereby decreasing the cost of the iron arsenate technology.

*Oxidation of arsenic trioxide in water and precipitation as ferric arsenate (Tran, 1991)*

*Oxidation:*



*Precipitation:*



Ferric arsenate is probably not a true compound, but rather an adsorptive binding of arsenic with ferric oxy-hydroxide, referred to as arsenic-bearing ferrihydrite ( $\text{FeAsO}_4 \cdot x\text{Fe}(\text{OH})_3$ ) (Robins et al., 1988; Robins et al., 1992; EPA, 1992). Surface spectroscopic methods, such as EXAFS (extended x-ray absorption fine structure), have been carried out on samples of arsenic-bearing ferrihydrite (Khoe et al., 1991) and reveal that adsorbed arsenic species are bound to surface ferric atoms by a coordination reaction (chemisorbed) (Khoe et al., 1991).

Although the arsenic-bearing ferrihydrite structure is considered to be stable over a number of years, in the longer term, the eventual crystallisation of the amorphous structure into goethite may slowly release arsenic in low concentrations to the environment (Robins et al., 1992; Harris and Krause, 1993). Robins et al. (1992) proposed this slow release of arsenic as a way to imitate the natural cycling of arsenic, gradually releasing arsenic back into the environment at low concentrations. Such a method for the controlled release of arsenic in concentrations below regulated emission levels could be considered a responsible long-term option for the management of arsenic waste.

## **9. Regulatory diections**

In the past, mine wastes were disposed of into tailings dams with little or no monitoring or safeguards to minimise leakage of heavy metals and other toxic compounds into the surrounding environment (Knapp, 1989; Sudbury, 1992). Considerable public concern in Canada, the U.S, Europe and Australia over contamination of land and water from heavy metals as a result of mining activities has led to greater regulation of emissions for heavy metals, including arsenic (Welch and Firlotte, 1989; Pearce, 1993). As a consequence, the mining industry has been forced to adopt more stringent controls on emissions and waste disposal of hazardous materials. Changes in technology and management practices for emission controls is particularly costly for industry in the shorter term. However, unless industry adapts quickly to the more stringent regulatory environment, the future costs of litigation will become potentially even more onerous for industry, epecially in terms of retrospective legislation for the past contamination of sites (Jaffe and Schoolman, 1990).

Anticipating increased regulations presents an important opportunity for the industry to significantly reduce its future costs of contaminant control. The introduction of improved environmental management practices (such as good documentation of on-site contaminants and the research of safe disposal techniques) over the longer term could reduce future costs. In addition, implementation of improved management practices by the mining industry will present it as a serious and competent land manager.

A number of regulatory issues are important to consider with respect to increasing the costs of control of arsenic emissions in the mining industry and will be discussed in this section. In particular, the decrease in emission levels following research showing arsenic to be carcinogenic and the increased liability for hazardous wastes, particularly retrospective legislation for the past contamination of sites, are two issues which will be considered.

### **9.1 Arsenic emissions**

The recently confirmed carcinogenic effect of arsenic on humans has the most serious implications for the mining industry in terms of producing potentially more stringent waste emission requirements. Government agencies in Britain, the U.S. and Canada are in the process of assessing the potential of low levels of arsenic to cause cancer. The likely revised guidelines reflecting this issue are shown in Table 9 (Pearce, 1993). The California EPA has introduced recommendations to decrease the maximum permissible concentration for arsenic in drinking water from 0.05 mg/L to 0.002 mg/L (EPA, 1992). According to Stöhrer (1991), these recommended levels are well below what has previously been considered a safe exposure level of 0.1 mg/L As (100 µg/L As) per day in drinking water (discussed in Section 3.2). Moreover, these

recommendations present fundamental difficulties for water and soil management in some geological provinces where background arsenic occurs in concentrations greatly exceeding the recommended arsenic levels (Table 1) (Pearce, 1993). Nevertheless, international regulatory authorities (such as the World Health Organisation (WHO)) are expected to adopt these arbitrary reductions in arsenic emission levels. Australia, largely without conducting its own research, will probably adopt the same recommended levels for arsenic emissions to the environment. Such reductions in emission levels for arsenic will have significant consequences for the mining industry (and government water authorities) in terms of increased costs of monitoring and the control of emissions.

*Table 9: Recommended maximum arsenic concentrations in drinking water (Pearce, 1993).*

<b>ARSENIC IN DRINKING WATER</b>		
<i>Britain</i>	<i>current 0.05 mg/L</i>	<i>recommended 0.01 mg/L</i>
<i>U.S.A.</i>	<i>current 0.05 mg/L</i>	<i>recommended 0.002 mg/L</i>
<i>Australia</i>	<i>current 0.05 mg/L</i>	<i>to be determined</i>

## 9.2 Hazardous waste legislation

U.S. and Canadian legislation for indications of future waste management policy are reviewed below because these countries have developed extensive legislation for the protection of the environment from mining and related activities, particularly with regard to liability (Alkire, 1984; Jaffe and Schoolman, 1990). In Canada, there are proposals to include arsenic as part of a list of 21 hazardous substances described as;

*...(the) most inherently hazardous due to their persistence in water or sediment, potential to bioaccumulate and toxicity..."*

(from Harris and Krause, 1993)

The only other inorganic contaminant on the list is mercury.

In Ontario, Canada, the stated purpose of a new programme for waste management (Municipal/Industrial Strategy for Abatement (MISA)) is to virtually eliminate the discharge of all persistently toxic materials from liquid effluents directed to natural water bodies (Welch and Firlotte, 1989). If regulations such as these become law, then the Canadian mining industry is faced with the possibility of having to phase out all emissions of arsenic to water and atmosphere. This is potentially a very serious situation, for there are very few mining operations where arsenic is not released.

Australia is a signatory of the Basel Convention which restricts inter-country transport of toxic wastes, including  $As_2O_3$  (Appendix 1, Kummer, 1992). Since certain

scrap metals, nickel catalyst and mineral concentrates theoretically fit into this category on account of the wide definition of "waste" and their heavy metal content, this legislation will undoubtedly restrict trade in metal recycling and recovery (Thompson Environmental Services et al., 1994). The Commonwealth Environment Protection Agency (CEPA) is in the process of reviewing Australia's waste trade for the Basel Convention legislation (CEPA, 1994). Once it has completed this review, it is likely to concentrate on domestic issues of waste disposal by introducing increased controls on disposal, for example of arsenic trioxide (pers. comm. Dr Anne Cawsey, CEPA, 1994).

Increased controls on emission and disposal of hazardous wastes are of concern to the mining industry for two reasons. The new regulation will result in higher control costs for arsenic emissions to the environment. In addition, the people constructing the policy for hazardous wastes at CEPA may have a limited understanding of mining waste management as they are usually not specialists in the field. It follows that the mining industry has an opportunity to improve communication and to educate policy makers of particular issues concerning mining wastes. This can be achieved by providing well-documented case-studies which deal with particular issues of concern. This approach would improve the knowledge base of regulators so that realistic guidelines can be developed. In Australia, the known human health effects due to arsenic exposure appear relatively minor due to limited use of groundwater for drinking. Environmental contamination by arsenic is mostly localised around industrial and mining centres (for example, in tailings dams). However, the potential health effects of exposure to arsenic in these areas may be serious and further research should be undertaken. Further research could put the relative dangers of arsenic contaminated land into context with environmental issues of broader impact, such as land/water salinisation and water quality of major river systems and groundwater.

### **9.3 Retrospective legislation**

In the U.S., legislation has been enacted because of the perceived ineffectiveness of existing federal statutes in dealing with the problems caused by the disposal of hazardous substances at abandoned sites (Alkire, 1984). CERCLA (Comprehensive Environmental Response Compensation and Liability Act 1980 (Superfund)) imposes strict liability on present owner/occupiers of the contaminated site, persons who owned the site at the time of contamination, or persons who arranged for disposal, treatment or shipment of hazardous substances at the site. Under current legislation, financial lenders who participate in daily activities of the operation are also liable for the cleanup costs (Jaffe and Schoolman, 1990; Williams, 1991).

The adoption of legislation similar to CERCLA in the Basel Convention (under Article 12) and in Australian legislation (such as the Victorian Environmental Protection Act 1970) has serious implications for the mining industry, particularly in terms of retrospective responsibility for contamination of the environment and from

increasing financial burdens through increased insurance premiums. Because of the nature of mineral recycling and disposal operations, it may be difficult to segregate the damage created by the activities of one owner and operator versus the activities of another. This difficulty may lead to a confused and complicated state of affairs resulting in increased litigation and ambiguity about liability (Tempkin and Tita, 1989).

It is important for the mining industry to anticipate the increased use of retrospective legislation in its management of contaminated sites (Mfodwo, 1991). Lenders, company directors and managers are increasingly targeted in Australian legislation for contaminated sites through a duty of care to the environment (for example, Section 62A of the Victorian Environmental Protection Act 1970). Australia and New Zealand are in the process of compiling guidelines for the assessment of contaminated sites for remediation (ANZECC, 1992). A number of states have published a register of contaminated sites possibly requiring remediation (for example, the Queensland and Victorian contaminated sites registers). Although still in the early stages of development, this trend is an indication of the future directions in legislation for contaminated land in Australia (and New Zealand).

## **10. Summary and discussion**

A broad spectrum of the implications of arsenic contamination and disposal from the perspective of both scientific and policy issues has been discussed in this section. This information is vital in order to anticipate government regulation and to develop future strategies for production and management of such hazardous wastes.

In summary,;

- certain species of arsenic are toxic and carcinogenic to humans at levels in drinking water exceeding 0.1 mg/L,
- the current maximum legislated concentrations for arsenic in drinking water is 0.05 mg/L,
- epidemiological research will possibly lead to further reductions in emission levels following the realisation that certain arsenic species are carcinogenic. Reductions for arsenic concentrations in drinking water to 0.002 mg/L have been proposed,
- the supply of arsenic currently exceeds the world demand,
- the production of arsenic as a by-product will increase as the mining industry continues to process more complex sulphide ores, such as low grade gold and arsenic-bearing nickel ores,
- as a result of the over-supply of arsenic, there is a problem of waste immobilisation for both existing and future stockpiles,

- retrospective legislation is likely to hold companies liable for past contamination of sites,
- future environmental regulation will result in increased costs for the mining industry for the control of emissions and the management of hazardous wastes,
- good documentation of arsenic production and environmental contamination is recommended at mine sites to improve management of metal wastes. This will provide policy makers with information of Australian conditions to develop better, more informed policy for arsenic emission levels and management of hazardous waste.

The problem of the contamination of land, water and the atmosphere by arsenic and other toxic metals is an issue that industry cannot afford to be complacent about. The uranium industry in Australia suffers from government over-regulation in response to perceived public fears of contamination. For example, the Ranger Uranium Mine in the Alligator River's Region of the Northern Territory is monitored by a group of about forty scientists and although largely funded by the Ranger Mine, is managed and controlled by the Commonwealth Government through the Office of the Supervising Scientist (Department of the Arts, Sport, the Environment, Tourism and Territories, 1989).

A similar response by government to heavy metal contamination represents a very real future burden for the mining industry, especially in light of increasing international control of hazardous wastes in the form of the Basel Convention (Appendix 1). In addition, with the increasing application of retrospective legislation, companies can now be held responsible for past contamination of sites. Thus, a conservative approach to contamination of the environment by heavy metals, employing well-considered management procedures, waste immobilisation and disposal techniques, is recommended.

As responsible land managers, the mining industry must document what it produces and emits in the form of hazardous wastes. This documentation may be required by government in the near future, and is already required for hazardous waste trade. If the mining industry can anticipate government regulation, then it is in a better position to control the formulation of new legislation and to minimise future costs and liability to the industry. Good documentation of contaminated areas and waste disposal will promote the development of trust between the industry, government and the general public, with the vital flow-on to land access and continued exploration and mine development.

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**12. Appendix 1:**

***Implications of hazardous waste legislation for the  
Australian mining industry : The Basel Convention***

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## *Implications of international hazardous waste legislation for the Australian mining industry: the Basel Convention*

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### 1. Introduction

The Basel Convention (*Basel Convention on the control of transboundary movements of hazardous wastes and their disposal*) came into force on the 5th May, 1992 in response to the growing awareness of the problems associated with the international trade in hazardous waste. The potential danger to human health and the environment from inadequate management of hazardous wastes are issues of legitimate international concern. Global exports of hazardous waste have increased over the past decade and it is estimated that industrialised countries produce approximately 300 million tonnes of waste per year (Johnson, 1991; Wallbaum, 1991; Williams, 1991). The Basel Convention represents the first major step in the comprehensive regulation of international transport and disposal of hazardous wastes.

The mining industry is a major source of metals in the environment and an active participant in the trade of hazardous wastes, particularly for the recycling and recovery of valuable metals. This report focuses on the implications of the Basel Convention for the Australian mining industry with a view to improving the industry's management of metals in the environment. First, the political influences on the development of the Convention are discussed because of the serious implications they hold for the future of the mining industry. Following this, an examination of the nature and extent of the obligations under the Treaty and the subsequent implications will be evaluated.

### 2. Background to the Basel Convention

The adoption of the Basel Convention represents the first global attempt to control the transboundary movement of hazardous waste (Kummer, 1992). Central to the Convention and presented in its preamble (and in Article 4(2)(a)), is the aim of waste minimisation at the source of generation as being the most effective way to protect human health and the environment. Where this cannot be achieved, the Basel Convention seeks to control and limit transboundary movement of wastes through the adoption of a comprehensive series of regulations to ensure that transport and disposal of wastes are conducted in an environmentally responsible manner.

Implications of the Basel Convention for the Australian mining industry, lie in the control and restriction of transboundary trade in hazardous wastes through increased regulation. A significant proportion of trade in hazardous waste consists of the recycling or recovery of valuable metals (and plastics, organics and paper) from such waste (Thompson Environmental Services et al. (TES), 1994). During the Conference of the Parties in March 1994, amendments to the Basel Convention were passed which restricted trade between countries and the banned all trade in hazardous

waste for recycling (Metal Bulletin, 1994). These particular developments are discussed in Section 3.2 of this report and are of major concern to the mining industry because of the loss of a valuable source of revenue. However, of equal importance to the future of the mining industry, is the political agenda of some organisations that aim to ban all trade in hazardous waste (Lipman, 1990; Wylie, 1992; Greenpeace, 1994).

While the overt aim of the Convention is to control the transboundary movement of wastes, its underlying intent is to prevent the dumping of hazardous wastes by the industrialised world on undeveloped nations (Johnson, 1991). A fundamental inequity exists between developed and undeveloped countries in relation to hazardous waste disposal. The increasingly stringent environmental regulation in many western countries, in addition to the scarcity of adequate disposal sites, has led to a significant disparity in the cost of disposal of hazardous wastes between industrialised and undeveloped countries. It is estimated that the current cost of disposal in the U.S. is \$2500 per ton, compared to \$2.50 per ton in some undeveloped countries (Lipman, 1990). Over the past twenty years, many industrialised countries have therefore exported waste for disposal in undeveloped countries rather than treat their waste domestically. Some disreputable incidents have occurred involving dumping of hazardous waste without informing the recipients of their contents (Wallbaum, 1991; Walsh, 1992).

Although the Basel Convention makes provision for recycling of hazardous waste under Article 4(9)(b), strong criticism has been delivered by Non Government Organisations (NGO's) such as Greenpeace, and also African nations who see this provision as a loophole for dumping waste on undeveloped countries (Wylie, 1992; Greenpeace, 1994). Disposal of toxic wastes in undeveloped countries is of concern primarily because these nations lack technological expertise and resources for monitoring or management. The possible contamination of land and water as a result of inadequate waste management can expose populations to toxic substances (Douglas, 1991). As a result, undeveloped countries have articulated their unwillingness to act as dump sites for waste produced by the industrialised world and are in favour of a complete ban in the hazardous waste trade (Schmidt, 1992; Wallbaum, 1991). Although it is of some concern to the mining industry, a discussion of the advantages and disadvantages of a complete trade ban and the possible implications for the mining industry are beyond the scope of this report. The final drafting of the Convention reflected a compromise between the interests of industrialised and undeveloped nations, which sought to control shipments of waste through a system of consent rather than a complete ban.

The Basel Convention is generally considered a positive step in the control of hazardous wastes and the prevention of dumping of these wastes in undeveloped countries (Walsh, 1992). However, there is strong and continued criticism of the Convention from those who consider the Treaty does not go far enough because it

does not prohibit trade outright. Equally strong criticism has been directed to the Treaty by the industrialised countries, particularly commercial sector interests who are concerned by the increasing controls on their trade in hazardous waste. Australian industry in principle supports the objective to control transboundary movement of hazardous wastes between nations and the adoption of environmentally sound control measures to prevent dumping (Australian mining industry position paper, 1994). However, the mining industry is especially concerned because of the potential impact of the Convention on trade in hazardous waste for recycling. It is important to appreciate the balance between the underlying intent of the Basel Convention and the political influences of concerned organisations discussed in this section, because these factors will ultimately impinge on the Australian mining industry.

### **3. Implications of the Basel Convention**

#### **3.1 Legislation**

The Basel Convention is regulated through a procedure of prior informed consent (Article 4(1)(c), Articles 6 and 7) which places the duty on the exporter to provide comprehensive details of the shipment of hazardous waste as outlined in Annex V(A) and (B) and to obtain the consent of the importing country and countries of transit, before export can take place. In addition, Article 4(8) requires the exporter to provide documentation to a relevant authority, that ensures wastes are managed by the importing country in an environmentally responsible manner.

Central to the Convention is the definition of "waste", which is given a very broad interpretation in the legislation. Article 2(1) defines a "waste" as:

*"substances or objects which are disposed of or are intended to be disposed of or are required to be disposed of by provisions of national law"*

Article 1(1)(a) defines a "hazardous waste" as:

*"Wastes that belong to any category contained in Annex 1, unless they do not possess any of the characteristics contained in Annex III"*

Annex I consists of two categories of wastes to be controlled. The first category lists wastes resulting from process streams, such as hospital, pharmaceutical and industrial wastes. The second category classifies wastes according to their toxic constituents, including substances such as lead, arsenic, zinc and organics. According to the legislation, if a waste includes any of the components in Annex I, the onus is on the exporter to prove that the wastes do not include any of the hazardous characteristics contained in Annex III, which include toxic, poisonous, flammable and explosive qualities.

Other provisions of the Basel Convention include:

- to trade with consent between member countries (Article 4(3), 4(5), 4(6), Article (9)),
- the establishment of a competent authority to administer the Treaty (Article 5),
- the development of a protocol for liability (Article 12),
- the establishment of a conference to consider development of the Convention and to transfer information between member states (Article 15).

Australia acceded to the Basel Convention on the 5th February, 1992 following the implementation of the *Hazardous Waste (Regulation of Exports and Imports) Act 1989* (the Act). This legislation resembles the Basel Convention primarily through its requirements of a cradle to grave documentation of waste movements and in use of the same classification scheme for hazardous wastes. The Act allows the export of hazardous waste only with ministerial approval obtained on receipt of a permit. However, the definition of "waste" in the Act is of concern to the Commonwealth Environment Protection Agency (CEPA), who is responsible for the implementing the Treaty in Australia (CEPA, 1994). The Australian legislation does not contain regulations for wastes with value and excludes wastes for recycling. This is in contradiction to the Basel Convention which makes no such discrimination. A change in Australian legislation to adopt the Basel definition will have a profound impact on Australia's trade in recycling wastes, as further discussed in the next section.

### *3.2 Major implications for the Australian mining industry*

#### *(i) Recent amendments to the Convention*

The most serious impact of the Basel Convention on the Australian mining industry arose as a result of the second Conference of the Parties to the Basel Convention on the Control of Transboundary Movements of hazardous wastes, in Geneva, on 21-25 March, 1994. This conference adopted the following amendments to the Convention:

*"-prohibit immediately the export of hazardous waste for final disposal from OECD to non-OECD countries and to*

*-phase out by 31 December, 1997 all exports of hazardous waste destined for recycling or recovery operations from OECD to non-OECD states"* (Department of Foreign Affairs and Trade, 1994)

Although the first decision has no immediate effect on industry in Australia as no exports of hazardous waste for final disposal are currently made to non-OECD countries (TES, 1994), the decision has a serious future impact in its potential to reduce the number of trading partners. The second amendment has a significant negative impact on Australia's trade in recycling and recovery industries and despite strong

lobbying by Australian industry and government to prevent the ban (Metal Bulletin, 1994; Wensley, 1994). Both amendments are of concern to the mining industry as they indicate a trend towards the total banning of trade in hazardous wastes.

A review commissioned by CEPA as part of a process to establish Australia's position in regard to the trade in hazardous recyclables found only a small proportion of Australia's trade was to non-OECD countries, primarily China, Malaysia, Philippines and India (CEPA, 1994; TES, 1994). Trade to these countries accounted for approximately \$22 million out of a total trade worth approximately \$120 million, primarily in non-ferrous metal recycling such as arsenic trioxide, spent nickel catalyst, lead batteries and in scrap iron-ore.

In many respects, the banning of trade in recyclables appears to contradict the concept of sustainable economic development (Brundtland, 1990), which supports recycling as a means to conserve resources for future generations. In addition, the original draft of the Convention included provision for recycling of hazardous wastes (Article 4(9)(b)). However, this argument ignores the reality faced by undeveloped countries who lack adequate resources and technology for managing the recycling industries required by western nations. The developing countries made a significant choice in their support of the current amendments to the Convention, in forgoing valuable revenue from the industrialised nations for environmental concerns. If the mining industry is to continue to play a role in shaping the future development of the Convention, then it must accept the strong environmental concerns of these developing countries.

Additional implications for the mining industry result directly from a number of inherent weaknesses in the documentation of the Treaty. The main areas of concern for the mining industry and which will be addressed include:

- the notice and consent system,
- objective criteria for classifying wastes,
- technical guidelines for determining environmentally sound management,
- liability of trading organisations.

#### *(ii) Notice and consent system*

The fundamental regulatory device for the Basel Convention is the procedure of prior informed consent (Article 4(1)(c), Articles 6 and 7). This system requires the exporter to compile extensive documentation relating to the nature of the waste for export (Annex V(A), Article 4(7)(c)), the environmentally sound management procedures to be used during transport (Article 4(2)(d)) and shipping details including packaging (Article 4(7)(b)) and transit countries (Article 6(4)) to be visited. Moreover, it is the responsibility of the exporter to document the environmentally sound management procedures to be used in the country of import (Article 4(8)). This includes procedures for recycling and disposal operations.

Documentation at this level is a daunting task for any organisation. In terms of the recycling industry, techniques for recovery will vary significantly depending on the nature of the product being recycled. Moreover, requests for information by the exporter regarding the importer's methods of operation may appear unnecessarily paternalistic and strain relations between the trading countries.

An additional implication for the mining industry are the substantial administrative resources required for the extensive documentation for export of hazardous wastes. These will result in increased costs, which may ultimately be borne by the consumer. The impact upon smaller companies and companies in less developed nations is likely to be greater as they have fewer resources at their disposal.

*(iii) Objective criteria for classifying wastes*

The Basel Convention fails in a number of important areas to clearly state its requirements through quantifiable guidelines and definitions. The definitions of "waste" and "hazardous waste" were presented in Section 3.1 of this report and contain no clear distinction between wastes destined for ultimate disposal in landfill and wastes which have value as a raw material for recovery operations or recycling.

In addition, the Convention contains no minimum concentrations or threshold limits for wastes to be classified as hazardous. Not only is it unspecific as to what material may constitute a waste but depending on the breadth of the interpretation, it may give rise to regulations that may impede or prevent the export of valuable products such as mineral concentrates. For example, copper concentrate containing trace amounts of naturally occurring arsenic or zinc may be defined as hazardous according to strict interpretation of the definitions. This situation is clearly unsatisfactory.

*(iv) Technical guidelines for the definition of "environmentally sound management"*

Article 2(8) defines "environmentally sound management" of hazardous wastes as:

*"taking all practicable steps to ensure that hazardous wastes...are managed in a manner which will protect human health and the environment...."*

Although this concept is crucial to the success of the Treaty, the vague definition and lack of any defined guidelines highlight a potential weakness. The mining industry fully supports the environmentally responsible management of hazardous wastes, however it requires detailed guidelines in order to comply. In addition, it is vital that waste management procedures are adequately monitored to determine their success and whether further improvements are required. The monitoring process also requires specific and well-defined guidelines to ensure its integrity.

Although the Basel Convention is primarily concerned with international trade in hazardous waste and is administered by the Commonwealth under section 51 of the

*Commonwealth of Australia Constitution Act, (1901)*, various aspects of the Convention present ramifications for State/Commonwealth relations. This is particularly with regard to ensuring the environmentally sound management of operations which generate or recycle wastes within state jurisdiction. However, a discussion of these ramifications and the possible impacts on the mining industry are well beyond the scope of this report.

*(v) Liability of trading organisations*

Article 12 makes provision for the adoption of a protocol for the field of liability and compensation for damage from the movement of hazardous wastes. The development of such a protocol could have serious implications for the mining industry in terms increasing financial burdens through increased insurance premiums. Liability which may be adopted include strict, joint, several and retrospective liability covering damage to property, impairment of the environment and loss of life.

It is important to monitor U.S. and Canadian legislation for indications of future waste management policy, as these countries have already developed extensive legislation for the protection of the environment from mining and related activities. Although the U.S. has not yet ratified the Basel Convention, certain aspects of its domestic legislation for hazardous waste regulation may provide an indication of the requirements that may be adopted by the Convention, particularly with regard to legal liability.

CERCLA (the Comprehensive Environmental Response Compensation and Liability Act 1980 (Superfund)) was enacted in the U.S. as a result of the perceived ineffectiveness of other federal statutes in dealing with problems caused by the disposal of hazardous substances at abandoned sites (Alkire, 1984). CERCLA imposes strict liability on present owner/occupiers of the contaminated site, persons who owned the site at the time of contamination, or persons who arranged for disposal, treatment or shipment of hazardous substances. Under current legislation lenders who participate in day to day activities of the operation are also liable for cleanup costs (Jaffe and Schoolman, 1990; Williams, 1991).

The adoption of legislation similar to CERCLA in the Basel Convention would have serious implications for the Australian mining industry, particularly in terms of retrospective responsibility for contamination of the environment. Because of the nature of mineral recycling and disposal operations, it may be difficult to segregate the harm created by the activities of one owner and operator versus the activities of another. This difficulty may lead to a very confused and complicated state of affairs resulting in increased litigation and not necessarily to the determination of liability for the contamination (Tempkin, 1989).

#### 4. Conclusions

The Basel Convention has several important implications for the Australian mining industry:

- (i) restriction in trade as a result of increased regulation recently exemplified by the banning of trade to non-OECD countries and the ban in recycling hazardous wastes,
- (ii) increased administrative resources required to document compliance,
- (iii) the lack of quantitative definitions and guidelines in the legislation which increases future uncertainty and risk for investors in the industry,
- (iv) increased financial burden through increased insurance premiums as a result of impending liability,
- (v) the underlying agenda of specific interest groups to continue to restrict the production of hazardous waste at the source of generation,

The mining industry has a unique opportunity to play an important role in the development of future protocols and legislation for the Convention and to safeguard its interest in the trade of hazardous wastes. To achieve this, the industry must appreciate and anticipate the environmental concerns of both specific interest groups and undeveloped countries for the environment and actively develop better communication with these groups. If the mining industry fails in this endeavour, the subsequent economic penalties may be substantial.

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