ABSTRACT

Molybdenum (Mo) and selenium (Se) were previously identified as constituents of concern in neutral mine drainage. This concern led to a review of the toxicity and management criteria for these constituents in aquatic and terrestrial environments. The characteristics of Mo and Se, including typical sources, environmental fate and transport, key toxicity effects, and reported thresholds for animals of interest will be reviewed in this presentation. Integration of these effects and environmental fate identify that assessments of toxicity will be complete if a site-specific risk assessment framework is used. An example of a pathways analysis for the fate of Mo will be considered to illustrate a complete site assessment. This information reveals the management criteria for Mo and Se should reflect site-specific environmental characteristics and the use of generic guidelines can result in highly conservative discharge limits for aquatic receiving environments.

INTRODUCTION

Activities at mine sites (mining, milling, road dust, waste management) all have the potential to generate environmental releases of a range of constituents directly to the adjacent aquatic and terrestrial environments. Management of such releases is accomplished, in part, through Canada’s Metal Mining Effluent Regulations (MMERs). These regulations authorize limits for most of the common constituents. However, there are a number of constituents associated with mine sites and mine effluents that are not currently addressed by MMERs, including the metalloids molybdenum (Mo) and selenium (Se). For these two elements, criteria and guidelines have been established for the protection of aquatic life (e.g., Canadian Council of Ministers of the Environment; CCME; CCME, 2007) and toxicity values identified for aquatic and terrestrial organisms. The absence of national regulatory limits for these elements reveals the need to identify management criteria and effluent limits for environments associated with mines. In this context, the priority elements identified in a review of neutral mine drainage from mines similarly included Mo and Se (MEND, 2004). One of the key recommendations made in the 2004 MEND report was to complete a detailed review of the toxicity related to exposure and site assessments for Mo and Se, as both elements are essential for animal nutrition (Barceloux, 1999a, b). As with many essential trace elements, any increase in environmental concentrations can have toxic consequences for sensitive species (e.g., Rand et al., 1995). This study addresses some of this past guidance, and follows some content and findings of a recent review of Mo and Se (MEND, 2008). This study contains a salient review of the characteristics of Mo and Se, including sources, environmental fate, transport and toxicity thresholds of relevance to freshwater habitats. A case study for Mo at a mine is used to illustrate these considerations.
Environmental Management Criteria

Most environmental management criteria for the constituents associated with mine site activities, including those in the MMERs, are based on an understanding of environmental concentrations that might cause some measurable degree of impact. This interpretation identifies the criteria as risk-based. Any risks from constituents like Mo and Se are generally considered in terms of frequency and extent of effect(s) on receptors in the environment. For such settings, the ecological risk assessment (ERA) process represents the accepted best practice to quantitatively compare levels of exposure against established toxic thresholds in most cases (Eisler, 1989; Hamilton, 2004; MEND, 2008). Typically, the ERA process involves successive tiers marked by decreasing conservatism and increased site-specificity. This process is predicated on the view any constituent that does not pose unacceptable risks in the first and most protective tier of ERA need not be considered further. Those constituents identified in the first tier as having potential to cause effects are then re-assessed with an increased level of realism or conservatism. The ERA continues until a satisfactory degree of realism and conservatism are achieved. If the constituent concentration exceeds a threshold, then management may be required (e.g., Lemly, 2004). A key challenge with the assessment of risk from Mo and Se is to adequately quantify both exposure and toxic thresholds for use in ERAs, particularly in the progression toward realistic assessments for mines.

METHODS

Information for this report was generated from a wide review of literature pertaining to Mo and Se, and was integrated with actual mine operations information for locations with differing geology across Canada. The review identified site-specific information for environments that range across the natural concentrations of Mo and Se, and had a focus on understanding the processes responsible for transport to the local and far-field aquatic freshwater environments associated with mine sites. For this report, the following major sources of Mo and Se information were reviewed: CCME; U.S. Environmental Protection Agency (EPA) toxicity databases; and World Health Organization (WHO). This review led to an understanding of the factors shaping Mo and Se toxicity that should be considered in the ERA process.

SOURCES AND TYPICAL ENVIRONMENTAL CONCENTRATIONS

Both Mo and Se are natural metalloid elements that occur widely in natural materials (bedrock, soil) and are usually present in freshwater and marine environments (reviewed by Driscoll et al., 1994; Goldberg and Forster, 1998). The Mo in some geological materials, especially along the Western Cordillera, allows primary extraction. Most Se arises via processing metal ores like copper, phosphates, and the production of sulphuric acid. Molybdenum is frequently present in the geologic environment as the primary metal sulfide, usually as molybdenite (MoS₂), in porphyry molybdenum deposits. There are also secondary forms of Mo, usually complexed with oxygen as metal-molybdates, in deposits dominated by copper, uranium, and zinc. Five oxidation states are possible although the +6 ion is most common. In water, the chemical form and availability of Mo varies with mixing patterns, pH, oxidation rates of organic sediments, reduction potential of sediments, and sedimentation patterns. For example, in circumneutral water that contains oxygen concentrations of at least 3 ppm, the common form will be the stable molybdate anion (MoO₄²⁻). By contrast, at pH of 3-5, molybdate frequently shifts to hydrogen molybdate
(HMoO$_4^{i-}$). The Mo in the low pH range, as typical of acid mine drainage (AMD) is usually adsorbed to sediment composed of clay or other oxic minerals. Also, the molybdate anion can be reduced to molybdenum disulfide or molybdenite (MoS$_2$) in low redox environments. In solutions with moderate to high concentrations of molybdate, different complex polymolybdate compounds will form. The major metallic Mo species include MoS$_2$ (the mineral usually mined to recover Mo) and ferrimolybdenite (Fe$_2$MoO$_4$). The accumulation and partitioning of Mo is favoured in sediments with high iron and organic matter, low redox conditions, and pH of 3 to 5.

Under most environmentally-relevant scenarios, Se usually occurs with other substances, such as sulfide minerals, including copper, silver, and uranium (reviewed by Simmons and Wallschläger, 2005). In water, sediment or soil, the Se usually binds with oxygen in inorganic forms that include selenate (SeO$_4^{2-}$) or selenite (SeO$_3^{2-}$); other forms include elemental selenium (Se$^0$) and selenide (Se$^-2$). In well-oxygenated waters, selenate is stable and the most common form under circum-neutral to alkaline pH. However, selenite will form in poorly oxygenated waters, and is less soluble and more reactive than selenate. The selenite will exhibit a greater partitioning from water to sediment relative to selenate. For stronger reducing waters, selenides may form, rendering the Se even less soluble and more sediment-bound. Selenium can also be found in the form of various selenium-sulfur compounds. Under acidic and reducing conditions, elemental Se (inert and insoluble) will occur, and with further reduction, selenides can form. It is possible that elemental Se will occur in soils experiencing AMD.

Anthropogenic activities like mining and agriculture that disturb natural materials act to increase environmental concentrations of Mo and Se. Prominent sources of Mo and Se include disturbance of soils with naturally high concentrations and burning of coal and petroleum (e.g., WHO, 2006). Other subtle sources of Mo and Se exist that are frequently excluded from ERAs. For example, atmospheric dry and wet deposition of Mo and Se is monitored in Canada (e.g., Gélinas et al., 2000). This atmospheric route has been identified as a minor source to terrestrial and aquatic habitats. However, evidence suggests that ash from forest fires can cause locally high concentrations of Mo and/or Se in surface waters such as lakes (e.g., Allen et al., 2003). A review (He et al., 2005) showed how Mo and Se arise from agroecosystems, and identified that adsorption-desorption, complexation, and precipitation-dissolution shaped transport to soils. Also, irrigation and fertilizer use represented other important sources (He et al., 2005).

In the absence of disturbance or forest fire ash, the soil Mo and Se content is usually determined by the parent material. Soils from North America show a range of Mo from 1 to 2 mg/kg whereas Mo-deficient soils show concentrations less than 0.2 mg/kg (Barceloux, 1999b). Soils with naturally elevated Mo and Se occur throughout western North America, (Nagpal et al., 1995; Frankenberger and Engberg, 1998). Groundwater concentrations of Mo and Se vary by location and attendant geology (Eisler, 1989; Frankenberger and Engberg, 1998). In natural rivers and lakes in Canada, the Mo ranged from 0.1 to 500 µg/L (Nagpal et al., 1995). In disturbed areas, the Mo of surface waters averaged near 70 µg/L. By contrast, Mo of the Laurentian Great Lakes ranged from 0.15 to 2.8 µg/L (Rossman and Barres, 1988). Concentrations of Se in surface water typically are very low compared with most other elements (i.e., <10 µg/L; Frankenberger and Engberg, 1998; WHO, 2006). In a Canada-wide survey of surface waters, Se ranged from <0.1 µg/L to 40 µg/L (Health Canada, 1992). The concentration of total Se in rivers and streams is typically less than 1 µg/L; however, concentrations near 200 µg/L have been measured, for
example, downstream of coal mines (Nagpal and Howell, 2001). Soils deficient in Mo or Se usually show low concentrations of these elements in adjacent surface waters (e.g., Eisler, 1989; Lemly, 2004). Also, Mo and Se are frequently elevated in municipal sewage sludge (Barceloux, 1999a,b). Concentrations of Mo and Se from Canada in treated mine waters released to surface waters varies from site to site, and Mo is usually less than 10,000 µg/L while Se is usually less than 10 µg/L (MEND, 2004; MEND, 2008).

ENVIRONMENTAL FATE AND TRANSPORT

The various forms of Mo and Se are frequently transformed by micro-organisms or macro-organisms. Almost all of the Mo and Se load that enters aquatic ecosystems in the dissolved form will, over time, either adsorb to particulate matter such as sediments and/or be taken up by biota (plankton, aquatic plants, invertebrates, fish; (Driscoll et al., 1994; Hamilton, 2004; Simmons and Wallschläger, 2005). Generally, the Mo and Se concentrations in water are positively correlated with concentrations in the sediment. Thus, any change in the Mo or Se concentrations in the water will be reflected by the sediment. However these changes in the sediment concentrations will not be instantaneous with changes in the water but occur over a period of months (e.g., Driscoll et al., 1994). The concentrations of Mo or Se in sediments are influenced by site-specific factors like flow rate, and other factors (e.g., organic content, pH; Howarth and Cole, 1985; Simmons and Wallschläger, 2005). Differences in temperature and precipitation across seasons are also known to influence fate of Mo and Se. For example, a species present in southern and northern locales will likely differ in response to Mo or Se exposure. This small and large-scale variation in Mo and Se fate also points to the need for site-specific studies (Eisler, 1989; Hamilton, 2004).

Generally, water-soluble Mo and Se can be readily taken up by biota in the environment. Studies have shown uptake of water-soluble Mo and Se occurs across cellular membranes and is correlated with exposure in water or consumption of sediment (e.g., Barceloux, 1999a,b). The solubility of Mo compounds also permits active excretion by organisms. For example, in fishes and mammals, the excretion of Mo readily occurs due to the action of the kidney, and the presence of high levels of sulfates and/or copper will enhance removal (Eisler, 1989). Also, the growth of some aquatic and terrestrial plants in media with high concentrations of Mo or Se results in high concentrations in all tissues (roots, stems, leaves, fruits); such species are regarded as hyperaccumulators (Eisler, 1989; Gao et al., 2003).

The transformation of Se to organic forms by organisms is referred to as methylation, and acts to increase the solubility and therefore bioavailability (reviewed by Simmons and Wallschläger, 2005). Generally, selenides can be the precursor to organic selenides (e.g. methylated selenides, soluble seleno-amino acids), and relatively insoluble inorganic metal selenides. In water, macrophytes and other plants (algae, phytoplankton) can readily take up selenite and selenate from water and incorporate Se in the tissue as selenomethionine. Anaerobic microbial reduction of selenate and selenite to insoluble elemental Se can represent an important mechanism for removing Se from water and transferring it to sediments (e.g., Lemly, 2004). Methylated selenides produced by biological reduction of selenite, usually occur at very low concentrations in water relative to the inorganic Se species, and differ between flowing (lotic) and standing (lentic) fresh waters (e.g., Simmons and Wallschläger, 2005). The accumulation patterns for Se from water to sediment identify higher rates of methylation in lentic compared with lotic environments, and correspondingly higher accumulation rates of Se in biota in lentic environments (e.g., EPA, 2004).
Studies have resolved the key factors that influence the fate of Mo and Se to different biota. These factors determine the degree that Mo and Se accumulate in plant and animal tissues, and includes:

- concentration and chemical form(s) of Mo and/or Se in the ambient environment;
- species and nature of exposure (e.g., life stage, natural and anthropogenic sources);
- media the biota reside and the diet and habitat use collectively determines the exposure route(s);
- chemical and other characteristics of those media (e.g., dissolved oxygen, pH, redox);
- timing and extent of exposure (e.g., short term, long term);
- abilities of organ(s) to remove Mo and/or Se from tissue; and
- presence of other constituents (e.g., arsenic, sulfate) that modify the bioavailability (see below).

TOXICITY EFFECTS AND ENVIRONMENTAL MANAGEMENT CRITERIA

The CCME (2007) guidelines to protect aquatic life in water identify 73 µg/L for Mo and 1 µg/L for Se. These guidelines reflect a range of past studies involving different organisms exposed to the elements in short (acute) and long-term (chronic) tests. It is useful to contrast these guidelines with observed toxicity thresholds for different aquatic species. Overall, the toxic effects of Se vary significantly among species and environments. As a general rule, the toxic potential of Se does not appear to change with chemical form. A review of acute studies (i.e., LC50) available from the EPA databases reveals the variability in the toxicity among species (MEND, 2008). These data suggest acutely lethal effects usually occur at Se of 500 µg/L. By contrast, lowest chronic effects occur from 2 to 5 µg/L, and denotes a threshold for concern of possible impact(s) on species in aquatic environments. However, these studies suggest that measurable effect(s) might not be seen at concentrations considerably higher than this, depending on the species and site-specific conditions (e.g., Simmons and Wallschläger, 2005). It also appears limits for Se have arisen from findings at problematic sites (e.g., Lemly, 2004). Studies of Mo toxicity also yielded variable results across species and environments. For most studies, the results to date indicate low risk from Mo to fish and invertebrates, and acutely lethal effects occur when Mo exceeds 100 mg/L (e.g., Davies et al., 2005) although Birge (1978) identified a threshold of 0.78 mg/L. Davies et al. (2005) also reported it was not possible to replicate the observations from the Birge (1978) study. The lowest chronic effect from Mo was more than 300 µg/L. Such disparity in scientific studies of Mo acute toxicity is the reason why British Columbia uses a Mo guideline of 2000 µg/L to protect aquatic life (e.g., Nagpal and Howell, 2001), and why the Mo toxicity to aquatic species was recently reviewed in the European Union (EU).

The new EU chemical legislation REACH has completed a qualitative evaluation of available acute and chronic aquatic toxicity data for Mo through the International Molybdenum Association (IMOa; www.imoa.org), using objective quality standard criteria (e.g., Klimisch et al., 1997). The outcome of this exercise revealed that very few studies of Mo toxicity could be considered as reliable. Consequently, this has led to the use of (unrealistic) high safety factors or the use of low-quality data for the derivation of water quality criteria (e.g., PNEC). Then the IMOa conducted extensive aquatic testing of Mo, to generate high-quality toxicity data for 10 species from 8 trophic levels. The tests ended in August 2008, and allow for the derivation of no-effect levels statistical extrapolation rather than the assessment factor approach (MERAG, 2006). This dataset – as it becomes available for regulatory purposes – will improve the scientific justification for (re)setting new or existing water quality criteria for Mo.
Because Mo and Se share chemical characteristics (e.g., stereochemistry) with other elements in the environment, the fate can be modified through interactions. Frequently, the interactions between Mo and Se and other elements lead to environmental antagonism, and this process reduces the actual compared with apparent toxicity to sensitive receptor species (e.g., Hamilton, 2004). For example sulphate shows similar stereochemistry with both Mo and Se, and Se is similar to arsenic and mercury. Generally, environmental antagonism due to stereochemical similarity represents a simple explanation why high concentrations of either arsenic or sulfate reduce the toxicity of Mo and Se (Rand et al., 1995). Also, the enzyme kinetics in exposed organisms determines partially the internal dose that acts on the site(s) of toxic action in the organism (e.g., Barceloux, 1999a,b). In combination, fluctuations in the environment, including seasonal cycles of precipitation, temperature, and metabolic rates synergistically shape the toxicity of Mo and Se. These processes further justify the simultaneous consideration of Mo and Se.

Regulatory limits that protect both aquatic and terrestrial biota associated with mine sites need to reflect the risk-based approach for receptor groups. Guidelines for effluent and environmental characteristics at mines are typically protective of aquatic life and/or drinking water quality (e.g., CCME, 2007). Although the aquatic ecosystem is the typical receiving environment, some scenarios exist where Mo and Se fate may necessitate consideration of risks to receptors of terrestrial origin. It is for this reason the CCME (2007) guidelines for irrigation water identifies 10 µg/L for Mo and 20 to 50 µg/L for Se. The guidelines identify risk of toxicity through the transfer via water particularly for Mo to some mammals. Specifically, mammals called ruminants (e.g., buffalo, cattle, moose) possess a multi-chamber rumen that allows for anaerobic digestion of plant material by microfauna. In the reducing environment of the rumen, thiomolybdate compounds are formed (e.g., Eisler, 1989). These thiomolybdates may bind with copper in the rumen, and cause copper deficiency. This process is termed molybdenosis and associated with symptoms involving reproduction, weight loss, and death (e.g., Frank et al., 2000). In Sweden, the discovery of molybdenosis was coincident with the widespread liming of lakes conducted to counteract the effects of acid precipitation that indirectly mobilized Mo across adjacent watersheds (Frank et al., 2000). In addition, it is likely that naturally low copper in some areas results in adaptation by wild ruminants and cause comparable patterns in moose. The risk of molybdenosis to moose associated with a closed mine is considered in detail as a case study to illustrate this complex process.

SITE ASSESSMENT STRATEGIES

For the assessment of Mo or Se from a mine site, the key first step is to resolve the form(s) of Mo or Se originating from the site in the air, dust, or discharge etc. It is then essential to resolve the contributions from the mine and the other sources (e.g., agriculture, ash from forest fires). To understand the potential for ecological impacts, it is necessary to then gain the best understanding of the receiving environment (e.g., flow rates, water volume, pH) that will determine the fate and partitioning of the Mo and/or Se. The next step is to resolve the aquatic and terrestrial species that will be exposed in the different environmental media and compare them with toxicity benchmarks. Such comparison of fate and benchmarks will elucidate the risk of effects (e.g., Eisler, 1989; Hamilton, 2004).
Studies were completed that assessed the potential for molybdenosis in moose associated with Brenda Mines, British Columbia. This mine produced copper and molybdenum from 1970 to 1990; closure and reclamation studies identified the potential for risk of Mo to ruminants on the site like moose, and so a detailed ERA was completed (e.g., Taylor and McKee, 2003). The basis for the identification of risk to ungulates at this site is that evidence indicates molybdenosis can occur as a result of exposure to aquatic sources of Mo in specific settings. For example, a mysterious moose disease in parts of North America and Sweden was eventually identified as molybdenosis and linked to exposure to Mo. These disease outbreaks have been reported in moose in places such as Manitoba (Gooneratne et al., 1989) and Sweden (Frank et al., 2000). These areas also have soil that is deficient in copper (e.g., Frank et al., 2000). Understanding the exposure pathways and risk from Mo to ruminants like moose requires a detailed analysis of the interaction between the animal and environment (e.g., Taylor and McKee, 2003).

Moose have two main sources of nutrition, a combination of leaves and twigs (referred to as browse) and aquatic plants (macrophytes) when available (e.g., LeResche and Davis, 1973). Generally, macrophytes only represent about 3% of the total diet, on a weight basis. Within the vicinity of Brenda Mine, surface water concentrations of Mo are usually less than 0.05 mg/L. The Mo from mine activities may be taken up by macrophytes and by riparian plants. Macrophytes includes water lily (Nuphar spp.) and pondweeds (Potomogeton spp.), and riparian species like alder (Alnus sp.), cattails (Typha spp.), sedges (Carex spp.), and willows (Salix spp.). The macrophytes will biomagnify Mo from water but with the short growing season (i.e., three months), it is unlikely they will accumulate levels found in terrestrial species grown on soil with high Mo.

Research at Brenda Mines suggests a typical Mo in macrophytes is around 332 mg/kg (BEAK, 2000). Because the Mo in browse is known to be below 1 mg/Kg or effectively zero, it is feasible to estimate the Mo in the total diet (BEAK, 2000). Thus, the weighted average of Mo in the diet can be calculated as: \[(6.6 \text{ Kg x 0 mg/Kg} + 0.22 \text{ Kg x 332 mg/Kg})/6.82 \text{ Kg} = 10.7 \text{ mg/Kg}\]. This average of Mo in the diet represents a value similar to the recommended critical threshold of 10 mg/kg in domestic cattle feed (O’Connor et al., 2001) and identifies little risk to moose. This interpretation can be viewed as conservative because the pathway of Mo to moose via macrophytes will be completely blocked during most winter months. The consequence is that macrophytes (plus sediment) will represents 90% of Mo consumption by a moose over 60 to 80 days per year, and water from the muskeg or lake, as the other 10% of Mo consumption, will be available for only about 150 days per year.

To understand risk to moose from Mo at the Brenda Mine site, the actual exposure period also needs to be known. For example, to ingest a significant quantity of Mo via macrophytes, a moose has to occupy the local area, the muskeg and/or lake, for an extended period of time (e.g., summer). A typical scenario of assessment for a moose would assume that an exposed animal would use the muskeg for 5% of an entire year, a lake for 20% of the year for a total of 25% of total feeding time. This habitat use estimate is conservative, as a single moose will typically range over 25 to 100 km² in ecosystems of the boreal forest. Also, this approach over-accounts for any focal foraging to obtain essential diet constituents (e.g., to seek salts, such as sodium chloride). Thus, the exposure to Mo is limited in areal extent, because the vegetation
(macrophytes or riparian plants) that contains elevated Mo from the mine will only be those plants in contact with water with elevated concentrations. Further, as soon as the intake of feed and water containing elevated Mo ceases, the Mo excess in the diet diminishes (i.e., excreted in urine) and any short-term effects are reversed (e.g., Eisler, 1989). This assessment predicted no significant adverse effect to moose via Mo release from a mine.

Field studies at the mine during 1999 documented the behaviour of moose. These studies confirmed the estimated exposure of moose to Mo in plants, and the absence of typical symptoms of molybdenosis (BEAK, 2000; Taylor and McKee, 2003). This analysis concluded the risk of Mo to moose was low. Over the last decade, no moose have been observed to suffer from molybdenosis, and the original assessment is interpreted as correct (G. Guilleminot, Site Manager, Brenda Mines, Xstrata Copper, Pers. Comm.). If the pathway for Mo was more complicated, it would be feasible to use a computer simulation package like IMPACT® software to quantify exposure and risk to local species. Such applications have previously been used to assess the risk of Mo to muskrat at other mines (e.g., Cameco, 2004).

DISCUSSION AND RECOMMENDATIONS

For both Mo and Se, studies identified species sensitivities vary across environments. The accumulation at low trophic levels (e.g., uptake into invertebrates and plants) has a great influence on exposure and potential impacts on animals at higher trophic levels. This process differs from biomagnification (i.e., a process that is associated with increasing concentrations of organic substances through three or more trophic levels through dietary uptake; Rand et al., 1995). Consideration of the toxicological profiles with Se reveals potential for food web impacts on predatory fish as well as birds and animals that feed on any class of biota (plants, invertebrates, fish) from the aquatic environment. Studies of Mo toxicity confirm ruminants are the most at risk animal group to effects whereas the toxicity to aquatic species like fish deserves reconsideration given the recent findings. Specific environmental management strategies for preventing such effects need to reflect key aspects of environmental fate and toxicity of Mo and Se. For example, a suite of remediation strategies have been identified to reduce transport of Mo and Se from mine sites. The planting of vegetation acts to reduce dust and disturbance of soils and movement to surface waters. Further, it is feasible to grow and harvest the roots and stems of plant species that hyperaccumulate Mo or Se, in aquatic or terrestrial environments (Barceloux, 1999a,b; Gao et al., 2003).

Management of Mo or Se across environments requires ERAs that quantify all exposure pathways of relevance. Such assessments for Mo are not likely to require the same level of detail and pathway delineation as for Se. This is primarily because Mo exhibits a lower rate of bioaccumulation than Se, and exposure via the food web of Mo is typically low relative to toxic thresholds. The role of irrigation water, as a vector that can move large concentrations of Mo and/or Se across habitats over short periods of time, always needs to be considered in ERAs. Generic default values of various transport parameters can be used as presented (e.g., EPA, 2004) but site-specific values are preferred. Also, an ERA should not be a stand-alone tool, especially for Se. Overall, the ERA should be a component of a full management plan that includes appropriate site analysis, and targeted and effective environmental monitoring programs.
REFERENCES


