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TREATMENT OF PIT LAKE WATER:

A REVIEW OF TREATMENT PROCESSES

November 2003

MINING AND MINERAL SCIENCES LABORATORIES
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Work performed for:

SRK Consulting Engineers and Scientists

Job No. 602722

MINING AND MINERAL SCIENCES LABORATORIES
REPORT MMSL 03-052(CR)

CONTENTS

Page

TABLES

FIGURES

APPENDIX

EXECUTIVE SUMMARY

On behalf of SRK Consulting, CANMET-Mining and Mineral Sciences Laboratories conducted a review of the treatment technologies for pit lake waters and assess the applicability of the treatment technologies to a northern climate such as that of the Faro and Vangorda pit lakes.

Development of treatment strategies for water contained in pit lakes is dependant on site specific needs. Each pit lake will have unique, geographical, meteorological, and environmental characteristic, and the chemistry in each pit lake is different. A mitigation strategy that is proven successful at one pit lake may not necessarily achieve the same water quality results at a different pit lake. The evaluation and design of associated measures for treatment of the pit lakes should consider factors such capacity, reliability, longevity, monitoring and maintenance.

The Faro and Vangorda pit lakes are located near the town of Faro, Yukon (Canada). The waters from these two pit lakes are slightly alkaline and contains elevated concentrations of Mg, Mn and Zn. Presently, the water quality in these two pit lakes is non-compliant and will need to be treated before being discharged into the receiving environment. Acid generation, at Faro and Vangorda mine site, will continue for many years. A treatment process for the waters from the two pit lakes must ensure long-term environmental protection and minimize the economic risks.

Chemical treatment has a long track record. It is robust and reliable and it is the technology of choice where the contaminant loads are high or there is no opportunity for experimentation However, depending on the volume of water that has to be treated from a pit lake, the cost may be high. Recent demonstrations of innovative technologies at Berkeley pit lake indicate that some treatment processes did not meet all the parameters specified in the objective for water quality. Based on the results of the demonstrations, the US-EPA record of decision was to use lime treatment to meet the objectives for water discharge from the pit lake.

Lime treatment is a proven technology that can remove dissolved metals, indefinitely, from large volumes of water and meet long-term environmental protection at these sites. However, while treatment with lime and ferric sulphate can be very effective, there are a number of drawbacks. These include the potentially high costs and challenges with sludge disposal and it may not meet permissible water discharge to the environment where pH, sulphate and/or hardness are limiting factors. Consequently, a number of mines are considering biological and other less-expensive treatment measures.

Biological treatment methods are a promising new technology for *in-situ* mitigation of pit lakes and the implementation and operational costs calculated from *in-situ* tests is considerably less compared to chemical treatment. However, adsorption of metals by phytoplankton and bacterial-mediated precipitation of metal as sulphides have not been demonstrated to be reliable *in-situ* treatment methods over the long term.

Site conditions play a major role in determining best management practices, and each site and procedure must be assessed on its own merits. However, there are also a number of generic information and design requirements. These include the following.

- **Location, Quality and Quantity of Contaminated Drainage**: Need to know volumes and loadings and for pit lakes, retention time, pO_2 and spatial changes within the water column.

- **Drainage Collection/Storage System**: Effective drainage collection is a critical component of a treatment system. If the treatment rate is limited, pre-treatment storage may be required to handle high flows. Post-treatment storage may be required for adequate dilution during discharge. A collection issue that may arise for pit lakes is contaminated drainage losses to groundwater if only the surface layer or overflow is remediated.

- **Treatment Process**: A treatment proposal should describe the proposed treatment process (chemical and/or biological reactions) and required conditions, such as pO_2, pH and redox, outline facilities, resource and management needs, predict post-treatment effluent quality and volumes over the range of predicted flows and input water quality, and indicate procedures for verifying the predicted performance.

- **Treated Effluent Discharge**: Requirements will depend on the effluent quality, quantity, discharge location(s), and authorized discharge limits and conditions.

- **Disposal of Secondary Waste Products**: Need to predict the quality, stability and quantity of any secondary wastes and a disposal plan.

- **Identify and Minimize Risk**: Measures to minimize the likelihood of failure include an ability to perform under extreme climatic conditions, comprehensive monitoring, frequent maintenance, well prepared contingency measures and having sufficient financial capability.

- **Capital and Operating Costs**: Along with effectiveness and reliability, existing and projected future capital and operating costs should determine which treatment strategy is selected.

RECOMMENDATIONS REGARDING PIT-LAKE TREATMENT AT FARO

Based on the limited information available, three processes of pit-lake treatment appear to be potentially feasible.

1. Lime or some other similar form of chemical treatment to raise the pH.
2. Amendments with sugar and alcohol to create anoxic conditions under which SRB precipitate metals.
3. Nutrient additions as means of stimulating the growth phytoplanktonic algae that remove metals such as Zn when they settle to the bottom.

Key questions (see previous section) with all three systems are:

1. At how high a metal load or flow rates can the system reliably meet permissible discharge concentrations, for how long and at what cost?
2. What is required in terms of process control, waste disposal, equipment, personnel, monitoring and maintenance, and discharge?

Lime treatment has been used successfully in the past for pit lake treatment at Samatosum in B.C. and is still used for lake treatment at the Levack mining complex in Ontario. Questions and concerns that should be addressed with potential lime use include:

- the lack of Fe to co-precipitate trace metals such as Zn;
- whether treatment should occur in the pit or only to the overflow;
- whether additional measures will be required to lower the resulting pH prior to discharge; and
- how to prevent discharge of the resulting precipitates.

Treatment of pit lakes with sugar, alcohol and proprietary amendments has been very successful in creating anoxic conditions under which SRB precipitate metals. A major limitation for SRB reactors is the input of oxygen in drainage or from surface processes such as wind action.

Questions and concerns that should be addressed regarding this strategy include:

- compatibility with water management limitations and requirements;
- oxygen inputs and the quantity and frequency and method of amendment application, and the resulting costs;
- how to sustain anoxic conditions, and deal with potential oxygen inputs from events such as rock fall, high runoff and high winds;
- measures required to address site-specific conditions; and
- how to address the issue of proprietary amendments.

Nutrient additions as means of stimulating the growth of phytoplanktonic algae, have been successful at least in the short-term both at Island Copper and at Landusky pit lakes. Successful results were also obtained from pilot-scale tests at Equity Silver mine. Equity has a cold climate that although not as harsh may be comparable to Faro. Questions and concerns that should be addressed regarding this strategy include:

- compatibility with water management limitations and requirements;
- the quantity and frequency and method of amendment application, and the resulting costs;
- measures required to address site-specific conditions, such as how to deal with seasonal constraints on biological activity; and
- how to address the issue of proprietary amendments (e.g., nutrients).

Temperature will be a concern at Faro, both from its potential impact on biological reactions and management. One way to limit this will be to treat and pump, treating the water during the summer and then pumping the pit lake down sufficiently so it can hold contaminated drainage during the rest of the year and there is no discharge until after the next treatment occurs. A potential downside is if pumping down the pit results in significant contaminant input from newly exposed mine walls.

Water management is a potential issue with all three treatment measures. The level of water in the pit lake can be lowered so that a large rock fall or mixing from wind action will not cause contaminated discharge to the environment. Another water management issue is the best place for dilution to occur. If diluted water still requires treatment, the extra volume can increase costs and cause other difficulties. Dilution may also cause dissolution of precipitated metals if they are at the solubility threshold, maybe reducing concentrations but increasing loadings. The ability to control uncontaminated drainage could therefore be quite important. Diversion of uncontaminated drainage away from unflooded benches will reduce contaminant additions to the pit lake.

Obviously more information is required regarding contaminant loadings, flow, retention time and oxygen inputs for a proper evaluation. Notably the data provided on pit lake chemistry did not include As, Mo, Se and Sb concentrations. Ferric sulphate is usually used to treat drainage where these elements are present in elevated concentrations.

The evaluation of treatment options for the pit-lakes also needs to consider the mitigation requirements for the site as a whole. The assessment of treatment alternatives for the pit-lakes should be expanded to consider the benefits of the mitigation requirements for the site as a whole, such as source controls, that might significantly reduce treatment costs. As for treatment of the pit-lakes itself, the evaluation and design of associated measures should consider factors such capacity, reliability, longevity, monitoring and maintenance. At Faro, it may be cost-effective to treat pit-lake water with drainage from the waste rock or tailings.

Lastly in an ideal world, a single treatment would result in the re-establishment of biota that would naturally remove contaminants to the required concentration. While this may be the final goal, in reality at most mines, responsible management includes regular inputs, replacement, monitoring and maintenance. The goal is how to most cost-effectively achieve conduct of this work. It is also important to note that there are no recipe mitigation strategies that are proven to work for all sites. Best management consists of the tools for developing the required site-specific understanding and mitigation plans.

ACKNOWLEDGMENTS

The author would like to thank Bill Price for performing the technical review of this report and David Koren and Doug Gould for their helpful comments. The authors would also like to thank Alexandra Lewis and Lidia Taylor for performing the online literature search.

TREATMENT OF PIT LAKE WATER:
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1.0 INTRODUCTION

Ground and surface water inputs following mine closure has resulted in pit lakes all over North America. Owing to the oxidation of exposed sulphide minerals on pit walls, and the leaching of soluble metals and acidity, many pit lakes are characterised by poor water quality (Miller, Lyons and Davis, 1996). Poor water quality in pit lakes raises environmental protection, regulatory compliance and long-term liability concerns. Consequently, considerable attention has been given to their management, characterisation and remediation (Miller, Lyons and Davis, 1996; Davis and Eary 1996; Levy *et al.*, 1996; Doyle and Runnells 1997; Atkin and Schrand 2000; Crusius, Dunbar and McNee 2002).

Remediation of pit lakes could involve either *in-situ* treatment of all the existing water or *ex-situ* treatment of a portion of the water. The best technology for the treatment of pit lake water and the cost associated with treatment will depend on characteristics of the lake, drainage input and output, and where water treatment will occur. It is essential to know water properties such as depth, spatial location, seasonal changes, chemical and thermal stratification, dissolved oxygen, sediment characteristics and biological activity before developing a treatment strategy (Doyle and Runnells, 1997).

The primary goal of post-closure plans for pit lakes where these factors are a concern is to reduce acidification and dissolved metal levels sufficiently to meet discharge permit levels. Chemical neutralisation of the huge volume of water can be very effective but cost can be high. Alternate technologies, which are less costly and environmentally acceptable, are being sought to remove the acidity and dissolved metals in pit lake waters. At the present time, enhanced *in-situ* natural biological processes that remove dissolved metals by means of adsorption by organic particles and sulphide precipitation and generation of alkalinity by means of Fe(III) and sulphate reduction are the most promising methods for pit lake remediation. These processes once established can be sustained through fertilisation and addition of organic material (if required) at minimal cost for long term.

The recovery of dissolved metal values from pit lakes could potentially offset treatment cost. The goal of many of the innovative technologies tested on water from Berkeley pit lake was to establish the feasibility of using oxidation and neutralisation, in combination or separately, to both accelerate and enhance the removal of dissolved metals and to assess recovery of valued metals such as Cu, Zn and Mn.

2.0 PROCESS USED FOR THE LITERATURE REVIEW

The open literature was searched for references on the treatment technologies for pit lake waters. The CANMET library carried out the search for the period 1980 through to 2003 using DialogWeb. A search of the open literature was conducted on twelve computerized databases using combination of key words listed in Table 1. These databases have index entries as recent as two weeks after publication and are assumed to have the most up-to-date information on water treatment technologies. A copy of the citations is included in the appendix.

The relevant papers were selected on the basis of the titles and information in the abstracts. Copies of the papers were obtained and reviewed and relevant papers cited in the selected papers were also obtained and reviewed. A search conducted on the Internet also produced several papers. The collected papers were reviewed for information on the following topics:

- application of technology to *in-situ* treatment of pit lake water;
- status of the technology (commercially available, pilot-scale, laboratory-scale and conceptual stage);
- effectiveness in removing dissolved metals;
- capital and operating costs; and
- sustainability of the effluent treatment process systems.

Table 1: Databases and keywords used for the literature search

Databases	
Georef	Mining Technology Database
Geobase	National Technical Information Service (NTIS)
SciSearch	SciSearch
Dissertation Abs Online	Mineral Processing Database
Wasteinfo	Enviroline
CA Search	Compendex Plus (engineering index)
Keywords	
Pit lake	treatment
open pit mining	remediation
adsorption	reclamation
bacteria	sorption
bioremediation	co-precipitation
decontamination	

3.0 PHYSICAL AND CHEMICAL PROPERITES OF FARO AND VANGORDA MINE PIT LAKES

The Faro and Vangorda pit lakes are located about 15 Km from the town of Faro, Yukon (Canada). Mining ended in January 1998. The sequence of mining was the Faro followed by the Vangorda and then the Grum pit. Tailings from the Vangorda and Grum pits were deposited in the Faro pit. There is no recorded backfill into the Vangorda pit. The region experiences a range in temperature from -46.1°C to 29.4°C; a mean annual temperature of -3.4°C; and a mean annual precipitation of 288 mm. The warmest month is July, which has a mean daily temperature of 11.5°C.

Water quality in the pit lakes is currently non-compliant for some metals. A summary of the water chemistry is shown in Table 2. Before the water can be discharged to the environment, the dissolved metal levels in the water leaving the pit must not exceed levels that would significantly affect the downstream environment. One of the closure options identified for these two pit lakes is to reduce metal loads entering the lakes and to re-direct flow from Vangorda Creek to Vangorda pit lake and flow from Faro Creek to Faro pit lake. Water flow from the lakes would then be discharged into the original creeks. This strategy may not be enough to achieve water quality objectives because metal sources from the pit rock wall and waste rock within the pit lake watershed would continue to enter the lakes. Consequently, water from the pit lakes could require additional treatment or some other form of mitigation before being discharged into the creeks (SRK Consulting, 2003)

Table 2: Water Quality of Faro and Vangorda Pit Lakes (data provided by SRK)

Parameters	Units	Faro		Vangorda	
				2 m	12.5 m
Conductivity	umhos/cm	1250	1190	444	1440
pH	s.u.	7.4	7.4	7.9	7.3
Acidity (to pH 8.3)	mg CaCO$_3$ eq/L	-	111	12	118
Alkalinity - total	mg CaCO$_3$ eq/L	-	197	66	82
Sulphate	mg/L	647	589	166	830
Aluminium	mg/L	0.1	0.42	,0.2	<0.2
Cadmium	mg/L	0.01	0.007	0.01	0.08
Calcium	mg/L	160	164	54	181
Cobalt	mg/L	0.05	0.08	0.03	0.41
Copper	mg/L	0.03	0.25	0.02	0.02
Iron	mg/L	0.14	0.1	0.06	0.11
Lead	mg/L	0.02	<0.01	0.05	<0.05
Magnesium	mg/L	60	55	17	70
Manganese	mg/L	2.8	3.0	1.5	20
Nickel	mg/L	0.09	0.06	0.05	0.41
Potassium	mg/L	15	18	<2	4
Sodium	mg/L	25	22	2	6
Strontium	mg/L	0.5	0.44	0.51	1.4
Zinc	mg/L	11.9	5.6	6.27	70.1

4.0 BIOLOGICALLY-BASED ACIDITY AND METAL REMOVAL TECHNOLOGIES

Numerous studies have been conducted on the use of sulphate-reducing bacteria (SRB) and algae for the treatment of and pit lakes waters (Dvorak *et al.*, 1992, Canty *et al.*, 1996, Steinberg *et al.*, 2001, Wendt-Petthoff *et al.*, 2002). Typically, biological remediation involves adding organic compounds or nutrients to pit lakes. Organic compounds have two functions, to deplete the dissolved oxygen in the water and sediment, and thus create suitable anoxic conditions for bacterially mediated Fe and SO_4^{2-} reduction, and to provide a carbon source for SRB cell growth. Nutrients are added to a pit lake to stimulate the growth of primary producers, mainly photosynthetic unicellular algae. The algae remove metals from the top water layer through absorption, adsorption and eventual settling to the lake bottom where the decomposition of organic material creates biological oxygen demand that may in turn create suitable anoxic conditions for SRB. Under anoxic conditions, SRB can reduce sulphate to sulphide through a series of enzymatic reactions. This process generates alkalinity, and can therefore neutralize acidity. In addition, sulphate reduction can precipitate dissolved metals either as sulphides in anoxic areas or from a rise in pH in overlying aerated zones. The cycling of iron from the sediment to the water column by SRB also contributes to the co-precipitation of dissolved metals

Fe(III) oxyhydroxide. Phosphorous and nitrogen sources may also be added, if required along with organic compounds to directly stimulate SRB activity.

Although SRB mediated processes occur in nature, their use in pit lakes to remove sulphate and heavy metals as sulphide precipitates and the eventual formation of stable minerals are relatively new and not proven effective for the long term (Benner *et al.*, 2000). SRB are presently being used in permeable reactive walls to remove acidity and dissolved metals, from groundwater (Benner *et al*., 1999).

4.1 Treatment of Acidic Pit Lakes in Germany

Eutrophication (increased nutrient supply) of pit lakes as a sustainable *in-situ* biologically based alkalinity generating processes is currently being assessed in Germany for the deacidification of pit lakes (Fyson *et al.,* 2003; Schultze *et al*., 2003; Vendt-Potthof *et al.*, 2002; Bozau *et al.*, 2000). These lakes have high concentrations of dissolved Fe and SO_4^{2-} and low pH values (Table 3). Vadeboncoeur *et al.*, (2001) investigated the effect of whole lake fertilisation on primary production in four small natural lakes in Michigan (USA).

Fyson *et al.* (2003) reported on tests carried out to investigate the effect of phosphorous and organic carbon addition on the water chemistry and biology of acidic lake water in the presence and absence of lake sediment. Phosphorus (200 µg/L) and organic carbon (acetic acid, 0.1 mmol/L) were added to PVC columns filled with 20 L of pit lake water. A layer of lake sediment (20 cm) was added to some of the columns. The columns were incubated at 18-22°C for 230 days. Addition of phosphorous and acetic acid, both in the absence and presence of lake sediment, managed to slightly elevate pH, and acidity was slightly reduced. Phosphorous concentration decreased rapidly through co-precipitation with Fe(III) oxyhydroxide and biological uptake. Addition of phosphorous and carbon did increased primary production, but acidity was only marginally reduced. The principal role of algal growth in the water column was to supply organic carbon and nutrients for the growth of SRB present in the lake sediment.

An additional treatment was tested in which 120 g of potatoes was added to each of the columns in the presence of lake sediment. Phosphorous concentration increased to a steady level of 100 µg/L for the two-year duration of the tests. The decomposition of the potatoes produced anoxic conditions for bacterial reducers of Fe and S species resulting in an increase in pH to neutral values. In a previous study, Fyson *et al.*, (1999) reported that anoxic conditions in the columns was temporary, but the near neutral pH and enhanced phosphorous concentrations were maintained for the two year duration of the test using potatoes as an organic carbon source.

Different organic materials such as mushroom compost (Dvorak et al., 1992) manure and sawdust (Drury, 1999) have been tested as an organic substrate for the growth of SRB in the treatment of acidic waters. Frommichen *et al.*, (2003) used 2 L enclosed microcosms to investigate the effect of different organic and inorganic carbon sources (lactate, ethanol, whey, molasses, carbokalk, etc.) in combination with wheat straw, on the microbial reduction of Fe and SO_4^{2-}. The best results were obtained with ethanol and carbokalk together with wheat straw. With wheat straw and ethanol, the pH increased from 2.6 to 5.5 and sulphate concentration decreased from initial concentrations of 11.5 to 13.5 mmol/L to final concentration of

<3mmol/L. A carbon source that is available locally can be used to reduce cost. In a lage-scale study, Koschorreck *et al.,* (2002) and Pohler *et al.,* (2002) investigated the addition of straw combined with carbokalk, a by-product from the sugar industry, as a means of stimulating microbial alkalinity generation *in-situ*, in an acidic pit lake. Straw bundles were placed at the bottom of an acidic pit lake and the physiochemical conditions and microbiology of the sediment-waters contact zone were studied. Addition of straw combined with carbokalk induced microbial alkalinity generation by means of Fe and SO_4^{2-} reduction in the sediment. Only Fe reduction occurred in the straw. However, according to Wendt-Potthoff *et al.,* (2002) to prevent loss of potential alkalinity by Fe(II) reoxidation in the water column suitable anions for Fe(II) precipitation as carbonate or sulphide must be present. In a similar study using carbokalk, Frommichen *et al.,* (2001) reported that pH increased from 2.6 to 6.5 in 1.25 m laboratory-scale mesocosm water column tests. In contrast, the pH and acidity in the water phase of the 6.4 m field-scale mesocosm were not altered compared to the untreated mesocosm. Hydrogen sulphide was measured in the water above the straw, even though there was no evidence to indicate that straw provides a good surface for bacterial film development. The authors concluded that the most important function of straw for pit lake remediation is as a long-term organic carbon source for SRB growth in the sediment. Neutralisation rates in both the laboratory and field mesocosms were calculated to be between 12 an 18 mol/m^2yr (Frommichen *et al*. 2001)

Table 3: Summary of some chemical parameters for acidic mining lakes in the Lausitz region in Germany. (Nixdorf *et al.,* 2001a)

Parameter	**Units**	**Mean**	**Range**
pH		2.9	2.4-3.4
Acidity	mmol/L	4.9	0.1-26.6
Fe	mg/L	95	0.2-800.0
Sulphate	mg/L	1448	460-4636
TOC	mg/L	3.4	0.8-10.9
Total N	mg/L	3.5	0.9-5.3
Total P	µg/L	14.3	4-26
chl a	µg/L	2	0.5-5.0

4.2 Test Work for the Gretchell Mine in Nevada

A laboratory investigation of the feasibility of the removal of dissolved metals was carried out using water from the Summer Camp pit lake in northern Nevada (Castro *et al.,* 1999). The Summer Camp pit lake is a shallow lake (18 m deep) located at the Gretchell Mine, in Nevada. The water is high in SO_4^{2-} and As, low in metals except for Fe, Na, Ca and Mg, and mildly acidic due to carbonate rock.

Two organic amendments were tested: waste from a potato processing plant and composted cattle manure. Lake water and sediment were placed in 5 L polyethylene bottles under anoxic conditions. The microcosm in the bottles received nitrate and phosphate with either the potato or manure amendment and the controls received no amendments. Initially, the addition of potato waste had the effect of slightly oxidizing the microcosms and retarding the onset of sulphate reduction. Addition of nitrate elevated the redox potential of the microcosms and prevented sulphate reduction because nitrate reduction is the preferred electron acceptor. A reduced nitrogen source such as ammonium salt is the better choice rather than nitrate. In most microcosms, pH increased from 6.1 to 7.

In general, once the SRB began producing sulphides, Fe concentration decreased to <1 mg/L and As concentration dropped to near zero. In these laboratory results, addition of approximately 200 g/m^3 of potato waste that contained 50% carbon as dry weight produced the best Fe, SO_4^{2-} and As removal.

4.3 Bio-Remediation Test Work in the Main Zone and Waterline Pit Lakes at the Equity Silver Mine

As part of a three-year research program, Lorax Environmental Services and the University of British Columbia, conducted field-scale tests to assess the feasibility of various bio-remediation strategies for elevated Zn, Cu, Ni, As and Cd in the former Equity Silver Mine near Houston, British Columbia (Canada) (Martin *et al.*, 2003; Crusius *et al.*, 2003; McNee *et al.*, 2003).

The Equity Silver Mine produced gold and silver from three open pits as well as underground workings up to its closure in 1994. Groundwater, surface run-off and precipitation were allowed to fill the Main Zone and Waterline pits. At present, the Main Zone pit lake measures 400 m by 800 m by 120 m deep. ARD neutralisation sludge is discharged periodically to the surface waters (~5 L/s) and sinks to the lake bottom. This causes mixing of the water column. As a result the water column has high dissolved oxygen levels down to the bottom of the lake (Pieters *et al.*, 2003). The highest total concentrations of metals, ~800 µg/L Zn, ~50 µg/L Cu and ~8 µg/L Cd, are associated with the sludge at the bottom of the lake. With the exception of a thin, <5 m deep surface layer created by runoff from the pit walls, dissolved metal levels (< 0.05 mg/L Zn, < 0.005 mg/L Cd and < 0.005 mg/L Cu) in the lake water from above the zone of sludge are low enough to permit discharge (< 0.2 mg/L Zn, < 0.01 mg/L Cd and < 0.05 mg/L Cu; (Mike Aziz of Equity pers. comm.). Strong stratification of the water layer during the summer months due to elevated temperatures in the surface water layer is the most likely cause for the concentration of dissolved metals in the top layer. The conductivity in the surface waters are significantly reduced during late August, most likely as a result of reduced water flow to the lake and mixing of the surface and deep waters.

The Waterline pit lake is 150 m by 500 m by 40 m deep. No sludge is discharged to the pit lake. The surface waters from the Waterline pit lake overflows into the Main Zone pit lake at a rate of 5-10 L/s during the summer. In contrast to the Main Zone pit lake, the water column of the Waterline pit lake is strongly stratified throughout the year and there is no significant biomass production, a feature attributed to the low phosphorous level. An oxic surface layer extends from the surface down to depths ranging seasonally from ~5 to 8 m. Below 8 m the waters are permanently sub-oxic and contains elevated concentrations of As, Zn and Fe (1.2, 0.8 and 45 mg/L respectively) (Whittle *et al.* 2003). In 2001, the concentrations of dissolved Cu and Cd near the surface were ~0.6 to 1.4 µg/L respectively. At 35 m depth, the concentrations decreased to ~0.2 µg/L Cu and ~0.1 µg/L Cd (McNee *et al.* 2003). The high levels of dissolved Fe and As in the sub-oxic waters, below 10 m depth (up to 1,500 µg/L As in deep waters), are much higher than in the fully oxidized Main Zone pit lake. The likely cause for the elevated As concentration is the dissolution of iron oxyhydroxides flocs that migrate down from the oxidised surface waters. (Crusius *et al.*, 2003; Martin *et al*. 2003).

Metal concentrations in the surface waters of both pit lakes are similar. This suggests that the sludge input into the surface layer of the Main Zone pit lake does not increase dissolved metal levels. Despite the high metal-rich particulates in the bottom water layer in the Main Zone pit lake, there is no evidence of dissolved metal enrichment. Preliminary evidence indicates that the pit walls are the primary source of dissolved metals in both pits.

Several limnocorrals experiments were conducted at the Main Zone and Waterline pit lakes to assess metal scavenging by biogenic particles and sulphide generation as a strategy to remove dissolved metals. Two nutrient addition strategies were tested using limnocorrals that physically isolated a column of water measuring 12 m deep by 1.8 m diameter. The two strategies were:

1. Addition of algal nutrients (phosphate and nitrate) to the surface waters of the Main Zone pit lake to stimulate algal production for enhanced metal scavenging by biogenic particles and subsequent particle settling; and

2. Addition of algal nutrients to surface waters and addition of dissolved organic carbon (ethanol) to deep waters of the Waterline pit lake to increase oxygen demand and foster the development of bacterial sulphate reduction and subsequent precipitation of metal sulphides.

Table 4 shows the timeline for the limnocorrals nutrient addition.

The limnocorrals were fertilised on a weekly basis with phosphate and nitrate at a molar ratio of 10:1. The limnocorrals in the Waterline pit lake, that received high doses of fertiliser, were injected with 500 mL of ethanol to speed up the onset of reducing conditions.

Phytoplankton production in the surface waters was greatly enhanced by fertilisation. The magnitude of phytoplankton production was more pronounced with the medium to high nutrient additions. In the Main Zone pit lake, the stimulation of algal growth from the medium and high nutrient additions decreased dissolved Zn levels at the lake surface from 250 µg/L to <2 µg/L. Total Zn levels in the Main Zone limnocorrals decreased to 30 µg/L (80% removal) and total Cu levels decreased from 5 to <1 µg/L. Cadmium and nickel removal were less pronounced. According to Reynolds and Hamilton-Taylor (1992), Zn is both assimilated by phytoplankton and adsorbed to the algal surfaces.

The addition of phosphorous and nitrogen in the Waterline pit lake limnocorrals greatly increased algal production in the upper 5 m of the water column despite relatively high levels of dissolved metals in the surface waters (e.g., 600 µg/L Zn). The addition of ~1 mmol/L ethanol at 2 m below the surface in Waterline pit lake limnocorrals produced reducing conditions and H_2S generation 7 m below the surface, one month after addition. Dissolved Zn levels in the surface waters were reduced from 600 µg/L to 150 µg/L over a two month period and dissolved Zn levels decreased by two orders of magnitude in the lower anoxic waters (Whittle *et al.,* 2003). The decrease in total Cd levels in the surface water (70% reduction) was attributed to particle scavenging settling and, in levels below 5 m, CdS formation. Copper levels, which were already low, were unchanged in both the control and treatment cells ([Cu] = 1 µg/L). A study conducted by Reynolds and Hamilton-Taylor (1992) in Lake Windermere, a natural lake in England, also noted that phytoplankton bloom in the spring had relatively little influence on Cu. The mass of Cu associated with the diatom bloom (*Asterionella*) in Lake Windermere was <5% of the total mass of Cu in solution compared to 50% of the total mass of Zn. The authors of the studies on the Main Zone and Waterline pit lakes and Lake Windermere speculated that Cu could be present in a species that is unavailable for uptake by phytoplankton. Iron, As and Ni concentrations were also unchanged in surface and bottom waters of the limnocorrals. The range of dissolved concentrations of Fe, As and Ni from 0 to 9 m depth were 0-13,000 µg/L; 0-500 µg/L; and 24-50 µg/L respectively. The authors suggested that the high dissolved iron concentration in the anoxic layer (as high as 30 mg/L) might obscure Fe removal as iron sulphides. Overall, the data collected in this study demonstrated that modest nutrient addition was effective in the removal of Zn and Cd, especially from the surface layer. Addition of ethanol to the deep waters was effective in enhanced H_2S production and the removal of Zn and Cd as sulphides. The results from the limnocorrals indicate that nutrient and carbon additions can cause significant improvements in water quality and may be effective for whole pit-lake remediation.

Table 4: Timelines for limnocorral treatment strategies

Pit Lake	**Event**	**Date**
Main Zone	Limnocorrals installed	24 June 2002
	Manipulation	
	• Controls: no nutrient addition	27 June to 30 September 2002
	• Low nutrient addition: 0.7 mmol P/m^2/wk	27 June to 30 September 2002
	• Medium/high nutrient addition: 1.4 mmol P/m^2/wk	27 June to 31 July 2002
	• 1.4 mmol P/m^2/wk	1 August to 30 September 2002
Waterline	Limnocorrals installed	10 July 2002
	Manipulations	
	• Controls: no nutrient addition	12 July to 30 September 2002
	• Medium/high nutrient addition: 1.4 mmol P/m^2/wk	12 July to 31 September 2002
	1.4 mmol P/m^2/wk	1 August to 30 September 2002
	• Ethanol addition (500 mL injected to the deep waters to two medium/high nutrient limnocorrals)	8 August 2002

Information on the removal of Cu and Zn by phytoplankton is also found in a study by Jackson and Bistricki (1994) of three lakes in Flin Flon, Manitoba. Lake Hamell, West Nesootao Lake and Schist Lake are contaminated with Cu, Zn, Cd, Pb, As and SO_4^{2-} from a smelter fallout. Schist Lake also receives the outflow from a tailings pond and municipal sewage effluent, and unlike the other two lakes, it is stratified and the bottom sediment is characterised by strongly reducing conditions (extremely low Eh), and high sulphide production and an ample supply of SO_4^{2-}. The results from energy dispersive X-ray and micro-analysis of plankton and lake sediments showed that all plankton samples contained measurable quantities of Cu and Zn, a few samples had detectable quantities of As and Pb and none had detectable Cd. Significant quantities of Cu and Zn were associated with FeOOH and MnOOH coatings on the exoskeletons of planktons. The FeOOH and MnOOH phases differ in their relative affinities for Zn and Cu. Zinc was selectively bound to MnOOH, whereas Cu and As were bound to FeOOH. Zinc and Cu were associated with Fe and Mn, suggests that Fe and Mn oxyhydroxides deposited on the exoskeletons had scavenged them from the water. The Cu, Zn, Fe and Mn concentrations on plankton exoskeletons and the concentration of bio-available Zn and Cu in the sediments were lowest in the lake whose sediments have low Eh and high free sulphide levels . Dissolved Fe and Mn released into the water column from anoxic but sulphide-poor sediments probably precipitated as FeOOH and MnOOH when they reached water with sufficient dissolved oxygen.. Strongly reducing conditions and H_2S production in the sediment interfered with the formation of both FeOOH and MnOOH because the Fe and Mn are sequestered as sulphides in the sediment . Therefore, Eh, sulphides and O_2 levels control the accumulation of metals by phytoplankton exoskeletons, as these factors control the precipitation of FeOOH and MnOOH. The authors concluded that a deep stratified pond characterised by anoxic bottom water, high concentration of SO_4^{2-}, low Eh values in the bottom sediments and high rate of production of free sulphide should be particularly efficient in precipitating and retaining heavy metals.

4.4 Bioremediation of Sweetwater Pit Lake

The open pit at the Sweetwater uranium mine, located in Sweetwater County, Wyoming was flooded in April 1983. Presently, the Sweetwater pit lake covers an area of 60 acres and hold 1.2 billion gallons of water. The dissolved selenium in the lake was 0.46 mg/L (above the reclamation standards of 0.05 mg/L) and dissolved uranium was 8.1 mg/L (above the livestock standard 5.0 mg/L). The lake contains approximately 1.2 billion gallons of water and it acts as an evaporative sink, thus the surrounding ground water is not affected. Kennecott Uranium Company hired Green World Science, Inc to use their patented *in-situ* metals immobilization for treatment of the pit lake. Green World Science, Inc. added 548.7 tons of sugars, fats, proteins, alcohols, phosphates and nitrates from October 19 to December 22, 1999 to enhance SRB activity already present in the lake.

The addition of nutrients dramatically accelerated biological activity in the pit lake. The dissolved selenium dropped below 0.05 mg/L on January 2001 and uranium was below 5 0 mg/L by March 2001. The average concentrations of dissolved selenium and uranium was 0.010 mg/L and 4.33 mg/L respectively, on October 2000. The cost of biormediation to date has been approximately US$0.05 per gallon (Paulson, 2001; Paulson *et al.*, 2002; Harrington, 2003). Unfortunately, these papers do not indicate predicted future treatment or monitoring costs or the annual oxygen input to the lake.

4.5 Revegetation of Mine Site and Wetland Plants as Passive Biological Treatment Method for Pit Lakes

Campbell and Lind (1969) investigated five coal strip mine lakes in Missouri. Three of the lakes were still acidic some 40 years after the end of mining, while the other two had attained neutral pH about 15 years after mining stopped. According to the authors, the main difference between the acid lakes and the neutral lakes was the remedial work at the different sites. The acid lakes received acid drainage from coal waste piles and had very little vegetation around them, whereas the two neutral lakes received drainage from forested areas and farmlands and have developed into lakes that resembled natural lakes with mildly eutrophic conditions. The factors potentially contributing to neutral conditions include organic input from the surrounding vegetation and reduced acid drainage through vegetative transpiration. The volume of fresh water flowing into each lake can also have a significant effect on neutralizing acidity. Unfortunately, the flows of fresh water into the lakes were not reported.

A somewhat similar situation was reported for the Hollister Mine site (Mudge *et al.*, 2003). Newmont Mining Corporation used a vegetation cover in the reclamation/closure of the 372-acres Hollister Mine site. This mine, which closed in 1992, is situated 38 miles northeast of Battle Mountain, Nevada. The closure plan required reclamation of the site for cattle grazing and wildlife. Precipitation in the region is approximately 12.3 inches per year. Summer temperature reach above 95°F and winter lows often reach –15°F.

The site consisted of two open pits, heap leaching pads, and overburden storage area. The heap leach and overburden was graded and re-sloped, then capped with topsoil and seeded. The reclamation/closure concept was to store the rainfall in the vegetated cover and later to release it through transpiration. This concept proved very successful, the heap leach flow rate decreased from 30 gpm in 1997 to 1.5 gpm in 2002, thus reducing the amount of water that needed to be treated. Of the two pits on the site, only one pit was below the ground water table (25 feet below). This pit had potentially acid generating material in the lower pit walls and on the bottom and potentially could degrade groundwater or harm wildlife. The company decided to create a natural cone of depression in the groundwater by allowing evapotranspirative loss from a wetland created in the pit. The bottom of the pit was backfilled with a layer of oxide waste rock amended with lime and limestone and capped with topsoil. The soil was sloped to keep the water that pooled away from the pit walls. The new pit floor was seeded and transplants of willow cuttings and wetland sod were introduced. Precipitation and groundwater inputs cause the water table to peak at about two feet above the new pit floor just after snowmelt. The results from the first three growing seasons indicate that the thriving wetland vegetation is capable of consuming the accumulated surface water by the end of July and keeps on evapotranspirating water through the growing season. The use of vegetative evapotranspiration to reduce drainage from the heap leach and create a sink for meteoric water and groundwater has proven to be a cost-effective strategy at the Hollister pit mine.

4.6 Island Copper Pit Lake

BHP Billiton Base Metals' Island Copper Mine (ICM) operated an open pit copper, molybdenum and gold mine on the north shore of Rupert Inlet, Vancouver Island, B.C. The closure plan for ICM involves the use of the 382 m deep pit lake as a semi-passive treatment reactor for five million cubic metres per year of acid rock drainage (ARD) from waste dumps. The initial plan proposed the development of an anoxic hypolimnion (bottom layer) for *in-situ* precipitation of the dissolved metals as insoluble metal sulphides that would then sink to the bottom of the lake.

In 1996 the pit was flooded with seawater over a 31-day period. Precipitation and runoff from surrounding area capped the seawater with a fresh water layer. ARD is injected at the 222 m level through two pipe. This created a three-layer meromictic pit lake. Table 5 presents a summary of the physical parameters of the pit lake (Wilton and Lawrence 1998, Poling *et al.*, 2003). The oxic top water layer is slightly saline (~3.5 ppt) and extends down to a depth of 7 m. This top layer, which is replenished by rainfall (~2 m/year) and weak ARD from mine walls and small masses of waste, must meet effluent permit limits before discharging into Rupert Inlet. A sub-oxic middle layer (2 mg/L dissolved oxygen) extends from 5 to 7 m down to a depth of 222 m. The low salinity injected ARD acts as a buoyant plume in the seawater rising until it reaches the surface layer. At the chemocline between the middle and surface layer, the plume spreads radially (Muggli *et al.*, 2000). A sharp pycnocline (salinity gradient 3.5 to 26 ppt) separates the top layer from the middle layer. As the plume rises through the middle layer, seawater entrained in the current may cause some mixing of the middle and upper water layers. A stable bottom layer (0.03 mg/L dissolved oxygen, salinity 28 ppt) has developed from a depth of 222 m down to the lake bottom at 382 m. Modelling has predicted that total anoxia will not develop in this senescent lower layer for another 10 to 20 years, although there are signs that it may be starting already (Poling personal communication). Table 6 and Table 7 show the chemistry of the ARD from two outlet pipes, discharge permit limits and chemistry of ICM pit lake water at different depths (Wilton and Lawrence, 1998; Poling *et al.*, 2003).

In 1996, after the end of flooding, temperature profiling of the water column showed a development of a distinct lower layer below the ARD discharge. The temperature, in this layer, is slowly rising (12.85°C). The temperature in the middle layer ranges between 12.3 and 12.4°C, and in the upper layer, the yearly temperature ranges between 4 and 20°C.

The initial plan was to add fish farm waste to the lake to consume oxygen in the lower layers of the pit. Some fish farm waste was added prior to and immediately after flooding. However, it was soon realised that a more cost-effective carbon source would be required (Price, personal. communication). The approach that the mine has subsequently taken has been to add nutrients to stimulate phytoplankton growth. From years 1997 to 2000, liquid fertiliser was distributed by boat to the surface layer every 10 days, to stimulate phytoplankton growth. Monitoring of the pit lake showed that phytoplankton-generated biological oxygen demand was not enough to consume the dissolved oxygen from the ARD flowing into the lake. A detailed fertilisation study in 2001 determined the effectiveness of fertiliser application on the precipitation of Zn, Cu, and Cd, and optimum fertiliser composition. From May 2001 to July 2001, 667 Kg N plus 74 Kg P (N:P = 9:1) was applied as a mixture of liquid ammonium polyphosphate (10-24-0) and urea ammonium nitrate (28-0-0), to the top water layer every 10 days. After July 2001, the quantity of fertiliser applied was reduced to 629 Kg N and 106 Kg P (N:P = 6:1) because of increased nitrogen level in the upper pycnocline.

Spreading liquid fertiliser every ten days throughout the top layer of the pit lake has been very effective in stimulating phytoplankton and bacterial blooms that, scavenge Zn, Cu, Cd and other dissolved metals. The pH values of the top water layer have been in the range of 8 to 10. Analyses of organic matter and inorganic sediment collected in sediment traps at different depths indicates that the mechanisms for removal of dissolved metals appears to be adsorption on planktonic exoskeletons and mineral precipitates, and subsequently sedimentation to the lake bottom. Dissolved Zn levels in the upper layer have decreased well below 0.2 mg/L. Recently, levels ranged from 0.01 to 0.02 mg/L Zn in the top layer (1 m depth). Zinc levels have also decreased in the middle layer. Fertilisation also removed most of the dissolved Cu from the top 1 m layer and maintained Cd levels to below the effluent permit level of 0.01 mg/L (Wilton and Lawrence, 1998; Poling *et al.*, 2003).

Monitoring of the ICM pit lake confirms that in-pit biological treatment of ARD can be successful in removing metals. The annual measured metal sedimentations reaching the bottom layer are: Zn, 33 tonnes; Cu, 59 tonnes; Cd, 0.97 tonnes. In addition to phytoplankton, Al and Fe oxyhydroxide precipitated from the ARD injected into the middle layer are believed to adsorb dissolved metals and settle to the bottom.

Diagenic processes in the bottom layer could also be cycling Fe and Mn from the sediment to the middle layer where they are then re-oxidised and settle out again. This process needs further investigation to determine its contribution to metal removal. The settling of phytoplankton blooms is presently maintaining the top-layer water well within permit limits for discharge to the environment. The cost of the fertiliser distribution into the lake and monitoring is presently about $100,000 per year. No value is available for the cost of the many on-going studies and extensive monitoring. Many of these studies address the concern identified when the system was first proposed, that the ARD plume would eventually mix with the upper layer, resulting in higher metal levels in the overflow that discharges to the ocean. The present studies include work to determine whether surface fertilization will lower metal levels sufficiently if some or all of ARD is applied at the surface of the lake.

Table 5: Island Copper Mine pit lake basic physical parameters (Poling *et al.*, 2003)

Physical Parameters	
Approximate total water volume of lake	241,000,000 m
Surface area of pit lake	1,735,000 m^2
Volume of top brackish layer (currently at 5 m depth)	8,680,000 m^3
Volume of middle layer	217,500,000 m^3
Volume of bottom layer	14,700,000 m^3
Volume of ARD flows into pit lake combined (1998)	4,500,000 m^3
Volume of ARD flows into pit lake combined (1999)	5,200,000 m^3
Volume of ARD flows into pit lake combined (2000)	3,440,000 m^3
Volume of direct rainfall on pit lake @ 1.88 m/y	3,260,000 m^3
Volume of run-in, not included in ARD estimate	500,000 m^3
Maximum depth	382 m
Total = ARD Flow	**3,760,000 m^3/y**

Table 6: ARD qualities and effluent permit limits at Island Copper Mine (Poling *et al.*, 2003).

	North Pipeline (mg/L)	**South Pipeline (mg/L)**	**ICM Effluent Permit (mg/L)**

pH	7.1-8.4	4.2-6.6	
SO4	400-780	1300-2000	6.5-11.5
Ca	140-230	280-410	
Mg	20-35	60-100	
Cu	0.010-0.040	0.030-3.3	0.05
Zn	0.5-3.9	4.2-11.1	1.0
Cd	0.010-0.030	0.010-0.050	0.01
Mo	0.008-0.012	0.0005-0.001	0.5

Table 7: Chemistry of ICM Pit Lake (Poling *et al.*, 2003)

	Top Layer	**Middle Layer**	**Bottom Layer**
DO (mg/L)	8-14	2.0	<1.0
Cu (mg/L) Nov-97	0.01	0.14	0.5
Zn (mg/L)	1.0		
Mo (mg/L)	0.5		
Salinity (psu)	~3.5	25	28
Salinity of ARD (psu)		1.0	
Temperature (°C)	4-20 (yearly range)	12.3	12.85
pH	6.3 to 10 over a 7 year period	6.8	6.5

4.7 Metal Adsorption by Ferric Oxyhydroxide

~~In the mining industry, f~~Ferric salts have been successfully used for the removal of As, Mo, and Se from effluents (Dening, 1996). These applications take advantage of the strong adsorption and coagulation properties of the insoluble ferric oxyhydroxides, which form in a neutral or slightly acidic media, as shown in the following equation (Riveros and Chaulk, 2000):

$$\underset{pH<0}{[Fe(H_2O)_6]^{3+}} \rightarrow \underset{0<pH<2}{[Fe(H_2O)_5OH]^{2+}} \rightarrow \underset{\sim 2<pH<\sim 3}{[(H_2O)_4Fe(OH)_2Fe(H_2O)_4]^{4+}} \rightarrow \underset{\sim 3<pH<\sim 5}{Fe_2O_3XxH_2O} \qquad (1)$$

Homestake Mining Co. ceased mining two open pits at the McLaughlin hot-spring type gold deposit in the Knoxville mining district of California, in 1996. Water was allowed to accumulate to a maximum depth of 85.3 m. The south pit is the more interesting of the two pits. The geochemistry of the south pit lake is shown in Table 8. In April 1997, the lake reached a depth of 68.6 m and had evolved into a stratified meromictic lake. A thermocline developed in the upper 7.6 m and pH decreased from 4.2 to 3.90 over the 7.6 m of the upper layer. The water was well mixed from 7.6 m down to a depth of 48.75 m. Water in the middle layer had a pH of 3.75, temperature of 13.6°C, conductivity of 11.24 mS and dissolved oxygen of 5.1 mg/L. A sharp chemocline spanning 1 m developed at a depth of 48.75 m. Anoxic water below the chemocline had a pH of 4.3, temperature of 12.6°C and conductivity of 9.65 mS. By August 1998, the lake reached maximum depth at 85.3 m. The bottom layer (below the chemocline) had increased in depth and the anoxic layer now extended to 12.2 m from the surface (Rytuba *et al.*, 2000).

The south pit lake receives water from hot springs located in the pit lake, ground water and drainage from ore stockpile and waste rock. Drainage flows from ore and waste rock piles are characterised by elevated concentrations of As, Sb, Ni, Hg. Cu, Co, Ni, Hg, Zn, sulphate and iron. The pH of the mine effluent has varied and ranged from pH 4.69 to 7.78. The hot springs near the bottom of the lake are slightly alkaline, pH 7.2 and input from drainage from the ore stockpile and surface runoff was initially alkaline, pH 7.51 because the drainage reacted with serpentine cut by carbonate veins in the rock. As the pit continued to fill, dissolution of the oxidized minerals on the weathered surface of the pit wall resulted in substantial increases in Co, Cu, Ni and Zn. These metals increased during the first part of 1998 when the lake was forming, and then decreased substantially in the latter part of the year, whereas Sb and As systematically decreased in concentration during filling of the lake (Table 8).

High concentrations of iron oxyhydroxide floc and dissolved iron (up to 158.6 mg/L total Fe) flow in to the lake. A plume of iron oxyhydroxide floc extends outward from the point of entry of the mine effluent into the lake. The Sb and As are apparently adsorbed onto the iron hydroxide floc. At the pH of the lake water (e.g., 3.97), As and Sb levels in the pit lake have remained relatively low (0.03 µg/L As and 2.7 µg/L Sb). Other metals, such as Zn, Ni, and Cu, are adsorbed more effectively at higher pH (Smith *et al.*, 1992; Aubé, 2003) and their concentrations have remained relatively high (Table 8). The author reported that the pH of the mine effluent varied between 4.69 to 7.78 during April 1997 to August 1998, and the spring water from the pit wall was near neutral, both flows are buffered by serpentine and carbonate veins, whereas the pH of the water in the South pit lake was acidic for the same period (pH 3.91 to 4.34). No explanation was given for the acidity in the lake, however it is expected that the upper layer of the lake will become less acidic as the pH is buffered by serpentine and shale and siltstones in the pit wall.

Mine effluent directed into the south pit lake has had very high concentrations of total and dissolved Hg, up to 79,505 ng/L and 279 ng/L respectively. Despite evidence that methylmercury is adsorbed onto iron oxyhydroxide, the total concentration of methylmercury has remained high, because of continued addition of effluent to the pit lake and favourable environment for mercury methylation in the lower anoxic levels of the lake. In the deeper water layer total and dissolved methylmercury concentrations are as high as 0.67 and 0.24 ng/L respectively.

Carbon dioxide released from gas vents is retained in the lower layer of the chemically stratified lake. The carbon dioxide continues to increase in the lower layer and this could lead to a limnic eruption, once the maximum capacity of dissolved carbon dioxide is exceeded (Rytuba *et al.*, 2000). The report did not discuss the potential effect of a release of carbon dioxide would have on the stratified layers and potential health hazard.

Table 8: Geochemistry of McLaughlin south pit lake water

Parameters	South pit lake					South pit mine effluent			
Sample Date	4/12/97	5/9/97	11/20/97	5/20/98	8/19/98	4/21/97	11/20/97	5/20/98	8/19/98
pH	4.34	4.08	3.91	3.7	3.97	7.51	4.69	5.93	7.78
Cond. mS	10.42	10.65	11.08	11.07	11.16	10.21	10.54	10.3	10.81
Temp. °C	16.9	21.5		18.3	26.1				
Fe mg/L	5.2	9.4	3.34	14.61	0.99	22	32.2	49.6	1.8
SO_4 mg/L	10.6	9.2	9.2	10.0	12.5	9580	8590	13,000	11,900
Zn mg/L	0.93	2.35	2.38	3.74	0.37	0.084	2.364	0.409	0.085
Cu mg/L	0.226	0.247	0.293	0.460	0.039	0.0022	0.341	0.0045	0.0001
Ni mg/L	19.0	217.0	28.2	33.7	3.0	7.2	25.0	11.8	0.47
Co mg/L	2.0	2.0	2.6	3.3	0.27	0.67	2.598	1.204	0.027
B mg/L	9.9	20.7	7.6	23.5	1.2	1.4	1.2	2.7	0.74
Pb µg/L	0.75	0.8	1.2	0.2	0.1	0.1	<1	0.1	0.1
As µg/L	23.3	40.9	14.9	9.51	0.03	9	33	5.44	0.03
Sb µg/L	28	47.6	15.1	8.8	2.7	39	24	3.5	2.9

5.0 DEMONSTRATIONS OF INNOVATIVE TECHNOLOGIES AT THE BERKELEY PIT LAKE

The US Environmental Protection Agency selected the Berkeley pit lake as a superfund site. In June 1991, EPA and DOE made funds available to support the Western Environmental Technology Office's operating contractor, MSE Technology Applications, Inc., and Montana Tech (University of Montana) for the development of the water treatment technologies for pit lakes.

The objective of the Western Environmental Technology Office was to support long term monitoring and characterization of several different aspects of the Berkeley pit lake system and demonstrate the effectiveness and technical feasibility of innovative remediation technologies on water from this and other pit lakes.

5.1 Berkeley Pit Lake

The Berkeley pit, located in Butte, Montana, is the largest pit lake in the USA (1,780 feet deep by over 1.0 mile in diameter). Since mining operations ceased in 1982, the pit has been filling with water at a rate of 0.75 to 1.0 million cubic feet per day. In fall 2002, the Berkeley pit lake was over 900 feet deep, contained 3.5 billion cubic feet of water and had a pH <3.0. The lake will eventually hold a volume of 9.1 billion cubic feet and have a surface area of 500 acres and a relative depth of 40% (Doyle and Runnels, 1997). Most meromictic lakes have a relative depth greater than 5% (Lyons, 1994; Anderson et al., 1985). Berkeley pit lake is not be expected to undergo seasonal turnover and should develop into a meromictic lake when the maximum depth is reached (Castro and Moore, 2000; Wetzel, 1983).

The deep water of Berkeley pit lake is separated from the surface layer by a chemocline approximately 35 to 50 feet below the lake surface. The hypolimnion has a pH of 2.5; temperature of 4.5°C; specific conductance of 8,600 µS; redox of 630 mV; and turbidity of 5

NTU. Dissolved-metal in the hypolimnion include Cu, 190 mg/L; Zn, 620 mg/L; Fe(II), 1000 mg/L; and As 2 mg/L. The epilimnion has a pH of 2.5; temperature of 0°C in winter to 25 °C in summer; specific conductance of 7,900 µS; redox of 670 mV; and turbidity of 30 NTU. Dissolved-metal concentrations in the epilimnion include Cu, 140 mg/L; Zn, 540 mg/L, and As, 0.21 mg/L. The ratio of Fe(II):Fe(III) is 2.5 in the hypolimnion and 0.36 in the epilimnion (Montana Bureau of Mines and Geology 2003).

An organic carbon source is required for growth of SRB bacteria. The amount of organic carbon in Berkeley pit lake varies from 3 to 5 mg/L. It is believed that a portion of the organic carbon in the Berkeley pit lake is from fuel and lubricant from machinery left in the mine. SRB activity was undetected in the pit lake in 1997. Earlier reports of SRB activity may have been in error, or they exist in isolated microhabitats. Furthermore, the lake bottom lacks anoxic black sediment that is an indication of good habitat for SRB (Berkeley Pit Water Treatment Research Project Mine: Final Report, April 1997).

The goal of many of the innovative technologies investigated in the Berkeley pit lake was to establish the feasibility of oxidation and neutralisation, in combination or separately, to remove dissolved metals and recover species such as Cu, Zn and Mn. The recovery of dissolved metal values from pit lakes could offset treatment cost. Innovative flow-though technologies for the removal/recovery of dissolved metals were tested at Montana Tech on water samples collected at a depth of 200 m in the Berkeley pit lake. The following are summaries of the test results.

5.2 Prevention of Acidic Drainage Generation from Open Pit Rock Walls

Open pit rock walls can contribute significant AMD flow into a pit lake, thus degrading the quality of the water and increasing the dissolved metal loading of the lake (Aubé 2003). As part of the research program of the US Environmental Research Agency's Mine Waste Technology Program, five innovative technologies for the reduction or elimination of AMD generation from pit rock walls were field tested (McCloskey *et al.,* 2003). Field demonstrations were performed at the Golden Sunlight Mine, located near Whitehall, Montana during July 22, 2002 to November 4, 2002. The *in-situ* spray spay that was applied included:

1. A modified furfuryl alcohol resin sealant, utilising wood and agricultural by-product to coat the rock surface;

2. Ecobond™ ARD is a phosphate-based AMD treatment process that reacts with Fe^{2+} to forms a stable, insoluble compound that coats the rock surface. This technology was developed by MT^2 of Denver, Colorado;

3. Magnesium oxide (MgO) passivation technology was developed at the University of Nevada, Reno Nevada. An inert coating is formed when a magnesium oxide solution contacts the sulphide rock. A pre-treatment wash is applied to the rock wall to raise the pH to greater than pH 11 before spaying the magnesium oxide solution onto the rock wall.

4. Potassium permanganate passivation technology was developed at the University of Nevada and DuPond (UNR/KP). An inert coating is formed on the sulphide rock surface with basic permanganate solution. It is hoped that this layer prevents contact with atmospheric oxygen, thus preventing AMD. This technology also requires a pre-treatment wash to raise the pH of the rock surface to greater than 12; and

5. Furfuryl alcohol resin sealant (FARS) technology was developed by International Polymers of Idaho Falls, Idaho. This technology consists of two-component, acid-catalysed binder that produces a stable resistant polymer.

Each compound was sprayed onto a 15.24 m by 15.24 m square of the rock wall by the vendors. Figures 1 to 3 show the total metal loading results for Cu, Zn and Fe from samples taken from mine wall-sampling stations. Table 9 shows the data for humidity cell tests for each of the treatment technologies that were conducted during 31 weeks period.

All test results indicate that the technologies can control the acid generation potential of mine rock wall and reduce the sulphate and metal loadings in a pit lake.

The FARS technology reduced sulphate concentration six times lower than the background concentration of 10,735 mg/L. Total loadings for Al, Cu, Fe, Mn, Ni and Zn were reduced by 4, 4.5, 3.3, 5, 8 and 17 times less than background concentration, respectively. For the humidity cell tests, Eh and pH increased slightly (pH range of 4 to 4.5). Over the 31 weeks, SO_4^{2-} was high with 26% of SO_4^{2-} resulting from sulphur oxidation and the mobility of Fe was substantial.

The Ecobond™ ARD reduced SO_4^{2-} concentration 3.7 times less than the background concentration. Total metal loading for Al, Fe, Mn and Ni were reduced below background. However Cu and Zn were greater than background concentration. A slight increase in Eh, Fe and SO_4^{2-} was noted. The final pH readings were greater than 7. Iron mobility was low, but SO_4^{2-} generation was high with approximately 15% resulting from sulphur oxidation.

The UNR/MgO treatment reduced SO_4^{2-} concentration by approximately seven times less than background concentration. Metal loading for Al, Cu, Fe, Mn, Ni and Zn was reduced by 2, 1.8, 1.1, 8.4, 2.8, and 5.2 times less than background respectively.

The UNR/KP treatment reduced SO42- concentration 3.6 times less than background. Metal loading for Al, Cu, Fe, Mn, Ni and Zn was reduced by 2.5, 5.0 1.5, 2.9, 4.7 and 5.7 times less than background, respectively.

A decrease in pH and alkalinity and an increase in Eh and SO_4^{2-} was noted at the beginning of the humidity cell tests for UNR/MgO and UNR/KP treatments. However, pH was greater than 6 at the end of the tests. No mobility of metals was noted with the exception of Mn. The low SO_4^{2-} generation indicates most of the SO_4^{2-} was not produced by oxidation of sulphide minerals but rather from sulphur oxidation.

Notably, none of these studies addressed the main concern; the longevity of the effectiveness of the coatings and the effectiveness of large scale field applications.

Figure 1: Total metal loading results for copper from samples taken from mine wall sampling stations (McCloskey *et al.*, 2003)

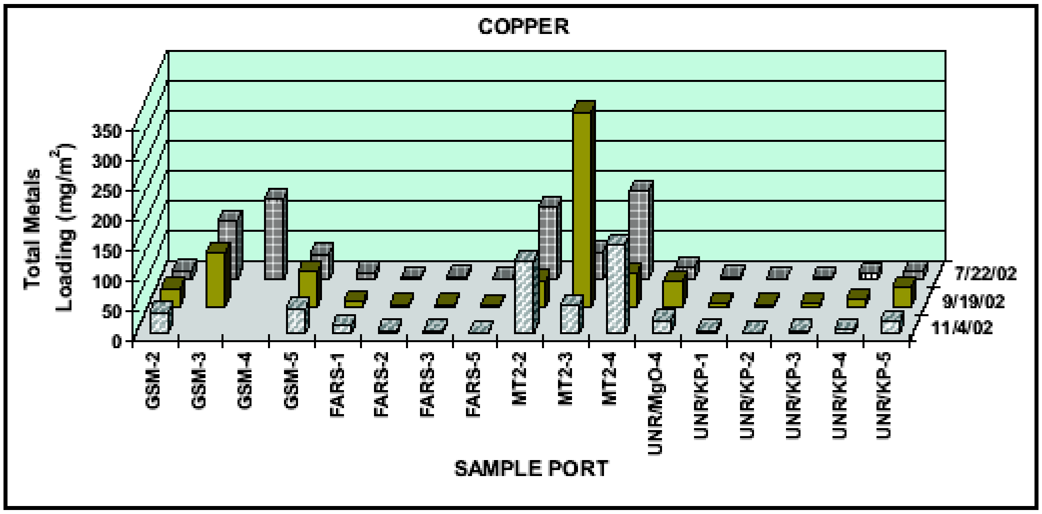


Figure 2: Total metal loading results for zinc from samples taken from mine wall sampling stations (McCloskey *et al.*, 2003)

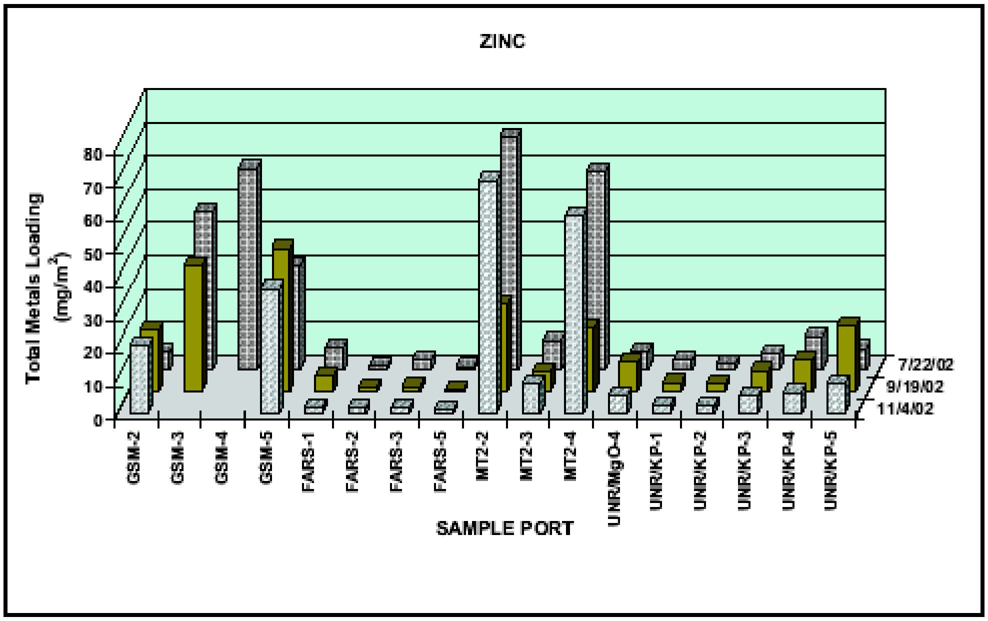
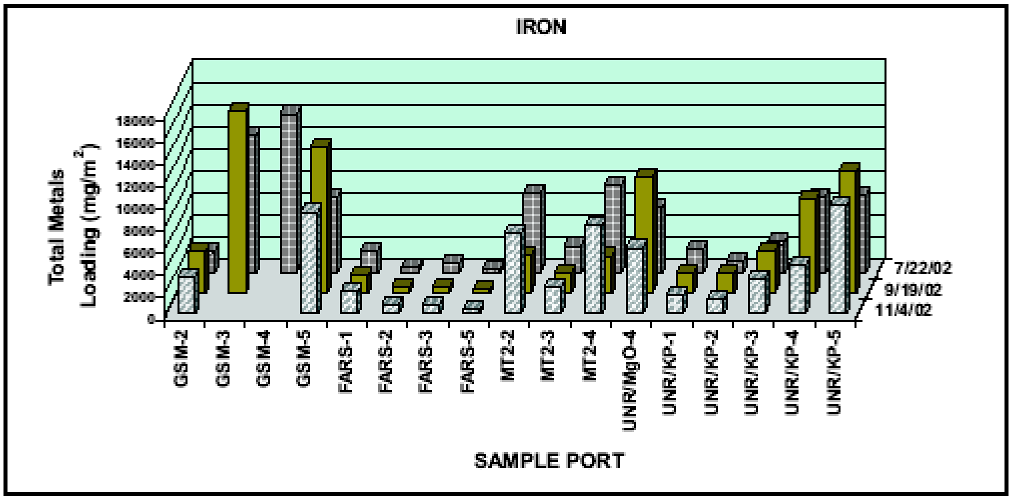


Figure 3: Total metal loading results for iron from samples taken from mine wall sampling stations (McCloskey *et al.*, 2003)

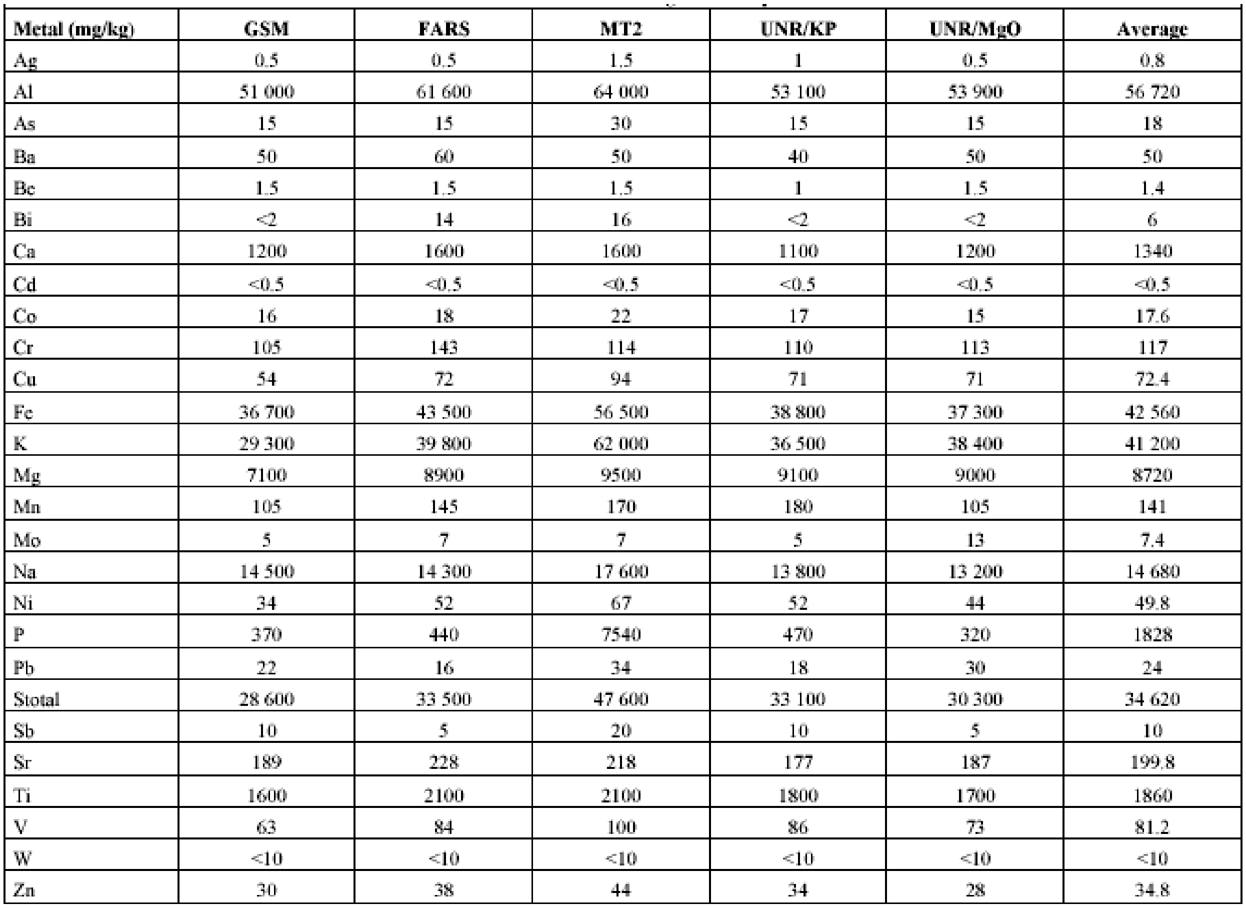


5.3 Azolla Biofilter

The *Azolla* biofilter process was developed at the Faculty of Agriculture, Hebrew University of Jerusalem, Israel for removing heavy metals from various industrial waste solutions. This technology is patented in the U.S. and in the European Union. A bench-scale demonstration of the *Azolla* biofilter system was conducted to determine the efficiency to reduce dissolved metal concentrations in Berkeley pit water using a single biofilter column and multiple biofilter columns in series. The demonstration was conducted in 1996 by SPC International, Inc., and The Hebrew University of Jerusalem (Berkeley Pit Innovative Technologies Projects: Final Report, August 1996 and March 1997).

The *Azolla* biofilter is produced from the biomass of the aquatic fern *Azolla.* In bench-scale tests, the *Azolla* biofilters removed effluent metals ions from Berkeley pit lake water to well below target concentrations (10 mg/L Al, 0.05 mg/L Cd; 0.05 mg/L Cr; 0.5 mg/L Cu; 10 mg/L Fe; and 0.02 mg/L Fe). From original water of pH 2.85, effluent pH levels increased to a range of 5.6 to 6.1. When two columns were connected in series, heavy metal concentrations were reduced even further. These test results proved that *Azolla* biofilter are capable of removing effluent metals ions to well below target concentrations. The cost of operation and capital costs of a treatment plant based on the use of the *Azolla* biofilter were not determined in this study. Operational conditions and engineering analysis of using *Azolla* biofiltration in a pilot or field-scale application for treatment of pit lake water have not been determined.

Table 9: ICP metal analysis results of treated and untreated rock wall samples in humidity cells (GSM represents untreated rock) (McCloskey *et al.*, 2003).



Metal (mg/kg)	GSM	FARS	MT2	UNR/KP	UNR/MgO	Average
Ag	0.5	0.5	1.5	1	0.5	0.8
Al	51 000	61 600	64 000	53 100	53 900	56 720
As	15	15	30	15	15	18
Ba	50	60	50	40	50	50
Be	1.5	1.5	1.5	1	1.5	1.4
Bi	<2	14	16	<2	<2	6
Ca	1200	1600	1600	1100	1200	1340
Cd	<0.5	<0.5	<0.5	<0.5	<0.5	<0.5
Co	16	18	22	17	15	17.6
Cr	105	143	114	110	113	117
Cu	54	72	94	71	71	72.4
Fe	36 700	43 500	56 500	38 800	37 300	42 560
K	29 300	39 800	62 000	36 500	38 400	41 200
Mg	7100	8900	9500	9100	9000	8720
Mn	105	145	170	180	105	141
Mo	5	7	7	5	13	7.4
Na	14 500	14 300	17 600	13 800	13 200	14 680
Ni	34	52	67	52	44	49.8
P	370	440	7540	470	320	1828
Pb	22	16	34	18	30	24
Stotal	28 600	33 500	47 600	33 100	30 300	34 620
Sb	10	5	20	10	5	10
Sr	189	228	218	177	187	199.8
Ti	1600	2100	2100	1800	1700	1860
V	63	84	100	86	73	81.2
W	<10	<10	<10	<10	<10	<10
Zn	30	38	44	34	28	34.8

5.4 Remediation of Berkeley Pit Lakes Waters Using Humic Material

Humic materials are produced by biological and chemical degradation of plant and animal matter. Chemical analyses of humic materials have demonstrated the presence of a large fraction of aromatic material and carboxylic acid and phenolic functional groups. These functional groups can form strong bonds with mineral and metal ions in aqueous solutions.

Experiments demonstrated that organic amendments could have a positive effect on the remediation of the water in the Berkeley pit. Of the four organic amendments tested (sawdust, aspen leaves, lawn clippings, and treated sewage sludge), the treated sewage sludge was the most effective at removing metal ions from the water and raising the pH of the acidic water (EPA/DOE Mine Waste Technology program, Annual Report 2001; MSE Technology Applications Inc., 2001).

5.5 Algal Remediation of Berkeley Pit Water

Research on the ecology of the pit lake systems has confirmed the great diversity of algae, fungi, and bacteria that live in Berkeley pit lake waters. Understanding the interactions of the different groups of organisms in a pit lake could indicate ways for bioremediation. However, ongoing research to determine the potential utilization of algae for bioremediation of the Berkeley pit lake has shown little promise. The algal *Chromulina freiburgensis* isolated from Berkeley pit water has been tested in the laboratory to determine its bioremediation potential. This algal did not remove Al, Cd, Cr, Cu, Mg, Mn, S, and Zn through absorption or adsorption from Berkeley pit water. Significant removal was detected for calcium (12.8%), iron (12.7%), nickel (8.4%), and silicon (56.2%) (Dakel and Mitman, 2001).

5.6 Neutra-Mill and Lime (CaO) Remediation of Anchor Hill Pit Lake at the Guilt Edge Mine.

The Neutra-Mill patented technology is being tested at the Gilt Edge Mine superfund site near Deadwood, South Dakota (EPA/DOE Mine Waste Technology Program, Annual Report 2001). I*n-situ* lime treatment of the 70 million gallons of acidic water (pH 2.7) in the Anchor Hill pit lake is using the Neutra-Mill technology, developed by Earth System, Pty. of Australia, combined with Green World Science’s patented organic mixture to stimulate the activity of indigenous SRB. This work is ongoing.

The Anchor Hill pit lake water contains high levels of Al, Cu, Cd, Fe, Se, Zn, SO_4^{2-} and NO_3^-. A Neutra-Mill adds lime to the lake to raise the pH to 7. It is expected that neutralisation of the water and stimulation of SRB will precipitate Fe, Cu, Al and other metal hydroxides. The Neutra-Mill is a floating platform containing an apparatus that grinds and mixes the lime into the lake. After neutralisation, the lake was allowed to sit undisturbed for several weeks to allow precipitated solids to settle and the system to stabilize before adding an organic mixture of methanol, molasses, and phosphoric acid to the pit lake (US-EPA webpage, 2003). It is hoped that increasing SRB growth should reduce or eliminate nitrate/nitrite and selenium, and decrease toxic metals concentrations to very low levels by precipitating them as sulphides. Bicarbonate was added to the water to increase alkalinity and to stimulate biological processes to help create a long-term, stable system. The pit lake will be monitored for several years to determine how well the two-step treatment work, how stable the pit lake water becomes, and the amount of re-oxidisation and remobilisation of precipitated metals.

Since the project was initiated in the spring of 2001, the metals of concern have decreased below closure standards for most regions of the lake (Arcadis webpage, 2003). Initial results indicate that neutralising the Anchor Hill pit lake consumed approximately 290 tons of lime. The lime efficiency was approximately 70%, with the excess lime settling to the bottom of the pit. The nitrate/nitrite and selenium concentrations have decreased. Biological treatment was proceeding slowly and sulphate reduction is not expected to occur until denitrification is complete (EPA/DOE Mine Waste Technology Program, Annual Report 2001).

5.7 Lime-Precipitation, Combined With High-Density Sludge Processes (HDS) Water Treatment Process At Berkeley Pit Lake

In less than 10 years, the water level in the Berkeley pit will approach the critical water level and a treatment plant will have to be ready to treat eight million gallons of water per day. The US EPA record of decision for Berkeley pit lake specifies that within eight years of approaching the established Safe Water Level (SWL) of 5,410 feet above sea level, design of the final water-treatment plant shall begin, with construction completed four years before the projected date when the water level will reach SWL (Watch News and Notes 2000).

None of the innovative technologies demonstrated for the treatment of the Berkeley pit lake water are developed to the stage when they can be implemented before SWL is reached. Considering the tight timeline in the contingency plan, Montana Resources and ARCO have started design work on building a conventional, two-stage high-density sludge lime precipitation process with aeration and polymer addition to remove dissolved metals from Berkeley pit lake water. The design will accommodate future expansion to the treatment plant for the integration of innovative metals recovery technologies and meet changes to water quality regulation for lower dissolved metal concentrations in the future. The huge volume of sludge generated in the plant will be deposited in Berkeley pit lake to minimize costs. Cost for construction of the treatment plant is estimated between US $12 to 22 million and operational cost is estimated to be approximately US $2 million each year.

5.8 Two-Stage Neutralisation Using Lime, Limestone and Sulphide

A two-stage neutralisation method was investigated to precipitate Fe(III), Al, and Cu from Berkeley pit water at a pH of approximately 6 (Berkeley Pit Water Treatment Research Project, Final Report, April 1997). After separating the precipitates from the water, the pH of the filtrate was raised to between 10.25 and 10.5 to remove the remaining metals, including Zn, Mn, and Cd.

The neutralisation process was able to remove metals to achieve the target effluent and secondary drinking water standards. Both Mn and Cd were below 0.001 mg/L. The pH of the first stage was kept at approximately 6. Lime and limestone were both effective, however limestone produced sludge with better settling and filterable characteristics. Aeration was necessary during the second stage of neutralisation to maintain the pH between 10 and 10.5. Waste materials, such as lime mud and reject lime grits, dust from a cyclone and fly ash, were also effective in neutralising pit lake water.

Sulphide precipitation was also tested as a pre-treatment and metal recovery process. Sodium sulphide ($Na_2S \cdot 9H_2O$) was added in the first-stage, and after separating the precipitate, lime (CaO) was added to the filtrate along with aeration to raise solution pH to approximately 10 to remove the remaining metals. A two-stage sulphide precipitation process was effective in removing both the Al and Mn to achieve the target effluent standards in Table 10. Over 99% of the Cu and Zn were recovered at the first stage at a pH of 4.3. Drawbacks to using sulphide precipitation and neutralisation include the large amount of H_2S gas that is released and the sludge generated from this process showed very poor filtration characteristics.

Table 10: Dissolved metal concentrations in Berkeley pit lake

Contaminant	Pit water (mg/L) Oct. 93 at 200 ft.	Secondary drinking water standards (mg/L)	Project objective (mg/L)

Aluminium	260	0.5 –2.0	1
Arsenic	0.8	NV	0.5
Cadmium	2.14	0.01	0.05
Copper	172	1	0.15
Iron	1068	0.3	1
Manganese	185	NV	2
Nickel	0.9	NV	NV
Chromium	0.078	NV	NV
Lead	0.098	0.05	0.3
Zinc	550	5	0.75

5.9 Green Precipitate (GP) Process

Geo2’s patented green precipitate (GP) Process is based on the formation of naturally occurring double cation hydroxy mineral compounds (e.g., pyroaurite, hydrotalcite, takovite, “Green Rust,” etc.) (Taylor *et al.*, 1998, Taylor, 1980) By bringing together in solution appropriate amounts of a divalent and trivalent metal, and adjusting the pH using lime, it should be possible to cause a large proportion of the metal cations in solution to react with one another to form a crystalline precipitate of Metal(II)Metal(III) hydroxy compound where the Metal(III) can consist of Fe(III) and Al(III) and various divalent cations contribute to the Metal(II) composition. The flowsheet for the process is shown in Figure 4.

The demonstration of the GP process had three objectives:

1. Demonstrate at a 30-L scale whether the precipitation of heavy metal species from Berkeley pit lake water will achieve target treatment results (mg/L in treated water): Al, 0.03–0.14; As, <0.04; Cd, 0.002–0.009; Cr, <0.04; Cu, 0.01–0.03; Fe <0.04; Mn, 5, SO_4^{2-}, 2500; Zn, <0.04; and pH 8.5–8.8.

2. Demonstrate whether leaching of the precipitate with a suitable reagent would recover at least 80% of the Cu and Zn.

3. Quantify the settling characteristics of the GP slurry.

In 1996, Geo2 of Melbourne, Australia, successfully demonstrated their GP process in a small laboratory-scale setup. The test demonstrated that dissolved metals in the Berkeley pit lake water could be made to react with each other under suitable conditions to form a precipitate consisting of members of the pyroaurite ($Mg_6Fe(III)_2(OH)_{16}.CO_3$) group of minerals. The average concentrations of heavy and toxic metals were significantly reduced and the average pH of the treated water was less than 8, making it suitable for agricultural applications.

In 1997, the GP process was demonstrated in a 40-litre batch reactor. The water was stirred at different rates under a blanket of nitrogen, and the dark green precipitate in suspension was removed through a tap at the bottom of the cylinder. The concentrations of dissolved metals in the treated water at pH 8 were: 0.006-0.016 mg/L Cd, 0.3 mg/L Al, <0.2 mg/L As, <0.05 Cr, 0.02-0.1 mg/L Cu, 0.02-0.1 mg/L Fe, 3.8–4.94 mg/L Mn, 0.1-0.05 mg/L Zn and 3,080-3,100 mg/L SO_4^{2-}. The concentration of Mn was slightly below the target level of 5 mg/L and SO_4^{2-} was above the target level of 2,500 mg/L. In tests conducted with a 500 L pilot plant coiled pipe reactor, the Mn level was generally below 1 mg/L. Dewatering the precipitate produced a compact filter cake. Leaching the precipitates with an ammonium carbonate solution, extracted 91.7% of the total Cu and 86.5% of the total Zn. Note that the concentrations of dissolved metals in the treated water are higher than the concentrations in the Canadian Environmental Quality Guidelines, which are 0.017μg/L Cd, 5-100 μg/L Al, 0.5 μg/L As, 2-4 μg/L Cu, 300 μg/L Fe and 30 μg/L Zn.

Compared to lime neutralisation, the GP process does not form scale of the surfaces and cause problems of blockages with pipes and other equipment in a thickener. The GP process provides an alternative to conventional lime treatment of pit lake waters (Taylor and Robins1998).

An economic evaluation based on the small laboratory-scale demonstration of the technology was not performed. However, the GP process demonstration did result in a 40% saving of lime compared to a demonstration of a conventional, two-stage lime precipitation process that was finally selected for the treatment of the Berkeley pit lake water by the US-EPA.

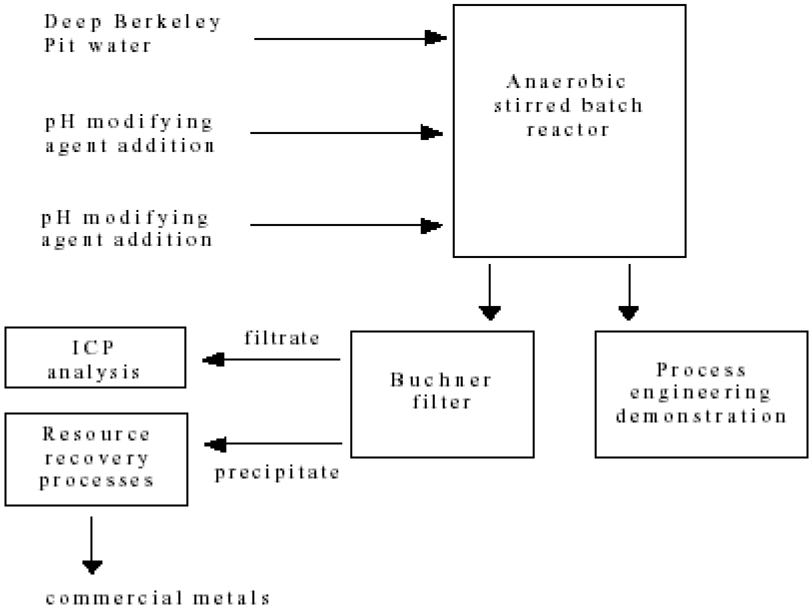
5.10 Kaolin Amorphous Derivative (KAD) for Treatment of AMD at Berkeley Pit Lake

Kaolin Amorphous Derivative (KAD) is a new suite of patented materials manufactured from low-cost clay minerals (MacKinnon *et al*, 1997; 1996a, 1996b). These materials show high cation exchange, high surface area and inherent basicity; properties that the manufacturer of this material hopes will effectively and economically reduce contamination from acid mine drainage.

The goal of this project was to treat water from the Berkeley pit lake to the target concentrations listed in Table 11. A flowsheet of the process circuit is shown in Figure 5. Tests were conducted at Montana Tech, University of Montana using on 200 mL samples of pit lake water, pH adjusted to 4.6 with $Ca(OH)_2$ and addition of 5.0 g recycled KAD (Montana Tech of The University of Montana Final, Activity IV, Project 7, Final Report 1999).

KAD material can be manufactured from kaolin and montmorillonite materials for specific adsorptive capacities for a wide range of heavy metals and concentrations in solution. The metal loading process takes place when potassium and sodium cations on the KAD are removed by exchanging with other metal cations such as Cu, Zn, etc. Metal loadings will vary according to the specific metal, water chemistry and properties that are manufactured into the KAD product. The metal ions can be stripped from the KAD for metal recovery and then KAD can be recycled back into the process. The estimated reuse of KAD is up to 20 times.

Figure 4: Batch version of the GP process



The bench scale trails on Berkeley pit AMD involved five steps as follows:

1. Pretreatment of Berkeley pit lake feed by oxidation with hydrogen peroxide and calcium hydroxide at pH ~4.6, to reduce the concentration of Mn and Al and adjust the Fe^{2+}/Fe^{3+} ratio.

2. Treatment with KAD material.

3. Stripping of the KAD material to remove specific metal ions.

4. Recycle KAD back into the process.

5. Final polishing with KAD to remove remaining metals down to target concentrations.

Hydrogen peroxide was used in the bench-scale tests to adjust the Fe^{2+}/Fe^{3+} ratio. Hydrogen peroxide would not be used in a full-scale operation, instead air sparging would be used.

Results from bench-scale tests of one type of KAD material demonstrated that, with the exception of Al, metal ions in a water sample from Berkeley pit lake can be reduced to the target concentrations and the target pH range for treated water (Table 11). Sulphate concentration was reduced by 50%, although the concentration in the treated water was still above the target level. According to the authors up to 50% of the available Cu metal as well as almost all of the available Zn could be recovered from the Berkeley pit lake water in this process. However, the report states that the data from the bench-scale tests implies that two or three-stage treatment with KAD columns may be required to reduce the metal concentrations to the target levels and maintain a pH of the treated solution at 8.5 or less.

Based on the amount of lime consumed during bench scale demonstration of KAD and the amount of flocculant added to a 13,000-gallons-a-day pilot-scale treatment plant demonstration, the daily reagent costs were calculated at US $23,200 based on 3-million-gallons-per-day of Berkeley pit pit lake water or US $0.77 per gallon.

Figure 5: Continuous circuit for AMD treatment and recovery of value metals using KAD

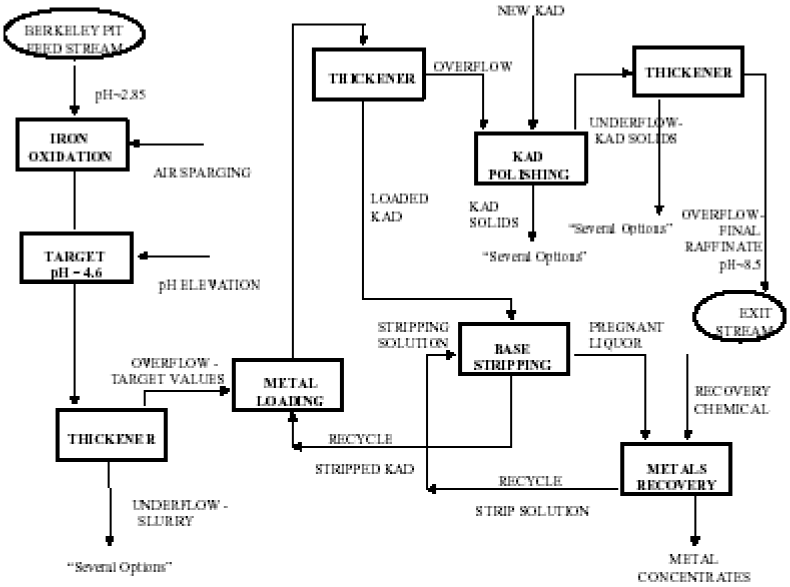


Table 11: Summary of metals concentrations (mg/L) for final polished Berkeley pit AMD samples using KAD

	pH	Al	As	Cd	Cr	Cu	Fe	Mn	Ni	Zn
Target value	6.5 – 8.5	0.05	0.05	0.003	0.05	0.01	0.30	0.05	0.10	0.10
Berkeley pit AMD	2.8	283	0.70	1.940	0.04	192	1020	229	1.08	611
Recycled KAD	7.2	0.08	<0.04	0.087	0.01	0.03	<0.01	40.9	0.02	1.53

5.11 Removal of Arsenic as Arsenatephosphateapatite Compounds

An extensive review of the literature was performed by Twidwell *et al*. (1999) on technologies and potential technologies for removing As from process and mine wastewater. Arsenic removal from wastewaters using lime neutralisation to form calcium arsenate compounds was the accepted industry method until the middle of 1980's. Lime treatment operated within the optimum pH range of greater than 10.5 is capable of providing a high percentage of As removal for influent concentrations of 50 μg/L, However, when in contact with carbon dioxide calcium arsenate converts to calcium carbonate with the subsequent release of As (Robbins, 1983)

The aim of the demonstration of As removal, as arsenatephosphateapetite precipitate, was to lower the As concentration to below the U.S. Drinking Water Standards of 50 µg/L (Montana Tech and MSE Technology Applications, Inc., Activity IV, Project 5, Final Report June 1998). The requirements for effective As removal from a solution asarsenatephosphateapetite and subsequent stability in tailings pond environments are:

- Arsenic must be in the arsenate rather than arsenite form.
- Phosphate is required for a stable, storable, solid product to be formed.
- The phosphorus/arsenic (P/As) mole ratio in the initial solution phase must be equal to or greater than 5.
- The Ca/(As+P) mole ratio in the initial solution phase must be greater than 1.5 times the stoichiometric requirement for formation of the apatite-like compound and for the formation of calcium sulphate.

This investigation demonstrated that apatite-like compounds, $Ca_{10}(AsxPyO_4)_6(OH)_2$, are formed through a phosphate precipitation process. Phosphate is added to the As bearing solution to form arsenatephosphateapatite compounds. These compounds have a very low solubility under tailings pond pH conditions, and they are more stable than calcium arsenate because As is chemically bonded to the formed compound, compared to adsorption of As onto ferrihydrate. The results of the demonstration are presented in Table 12.

Table 12: Effectiveness of arsenic removal by phosphate.

	Initial arsenate concentration µg/L	Arsenic after treatment, µg/L		
		P:As = 0	P:As = 5	P:As = 7
Pure water	1,000,000	58	< detection limit	--
Berkeley pit lake water	100,000	14	14	< detection limit
ASARCO water	>3,000,000	2,776	19	24
The mole ratio for pure water Ca:(As +P) = 3.7; Berkeley pit lake & ASARCO water Ca:(As +P) = 2.5				

The long-term stability of the apatite-like compounds formed in the demonstrations were tested by air sparging into aqueous/compound slurries. The six-month data for all samples showed an As solubility of less than 50 µg/L. Stability test monitoring was continued for 18 months. The final test result was not available for inclusion in this report.

A preliminary economic analysis of the apatite-like precipitation process estimated the treatment cost for low As bearing water such as Berkeley pit water is approximately US $0.74 ±0.21 /1000 gallons. The estimated cost for treating a very high arsenic bearing wastewater was estimate to be US $0.175 ±0.5 /gallon (Table 13).

Table 13: Cost analysis for removal of arsenic as arsenatephosphateapatite compounds

Conditions	Operating cost $/yr	Total cost $/yr	Cost $/1000gal
300 gal/min, 500 µg/L As	27,600	118,400	0.74 ±0.21
20 gal/min, 1g/L As	143,700	184,300	17.5 ±5.2

5.12 **Removal of Selenium**

Three technologies were selected for field demonstration of Se treatment/removal alternatives by the U.S. Environmental Protection Agency's Mine Waste Technology Program (EPA-MWTP) (MSE Technology Application Inc., Activity III, Project 20, 2001).

The technologies were:

1. ferrihydrite precipitation with concurrent adsorption of Se onto the ferrihydrite surface (ferrihydrite adsorption) optimised by MSE (EPA's Best Demonstrated Available Technology (BDAT));
2. a catalysed cementation process developed at Montana Tech of the University of Montana; and
3. a biological Se reduction (BSeR™) process developed by Applied Biosciences Corporation (AB) of Salt Lake City, Utah.

These technologies were tested at the Garfield Wetlands-Kessler Springs and the target concentration of total selenium in the treated effluent was <50 µg/L. Ferrihydrite adsorption is considered EPA's best demonstrated available technology. As such, ferrihydrite adsorption was used as the baseline technology for comparison with the innovative Se removal processes. All three of the processes were able to achieve the target level for Se under optimised conditions. Table 13 shows the results from the field demonstration for each technology and also includes results from additional testing of the catalysed cementation process.

The BSeR™ process performed most consistently during the demonstration. During the 187 days of evaluation, all but four effluent samples from the BSeR™ process were below 10 µg/L, and greater than 70% of the effluent samples were below detection of 2 µg/L. Presently, Kennecott Utah Copper Corporation (KUCC) is recycling 100% of the Garfield Wetlands-Kessler Springs process water back into the operation as makup water. KUCC has identified the BSeR™ process technology as the favoured treatment for Se at their site, when all the water flow cannot be recycled into the operation.

An economic analysis of the processes to treat 300 gpm flow at the Kessler Springs site was performed. The retrofit of a vacant water treatment plant and associated equipment at the Kessler Springs site was used as the basis for the capital costs. Table 14 is a summary of the economic analysis for the selected technologies treating groundwater with 2 mg/L Se operating at 300 gpm. The BSeR™ process was the most economically attractive technology demonstrated during this project. The figures represent an order of magnitude cost estimate of the total net present value for each process

Table 14: Demonstration results summary selenium removal (MSE Technology Application Inc., Activity III, Project 20, 2001)

	Ferrihydrite Adsorption Results	
Treatment Condition	Mean Selenium Effluent Concentration ± Standard Deviation (n = sample size)	Minimum Selenium Concentration
Low iron (~1400 mg/L iron)	304 µg/L ± 69 (n = 27)	115 µg/L
Medium iron (~3000 mg/L iron)	201 µg/L ± 103 (n = 13)	42 µg/L (at midpoint of process)
High iron (~4800 mg/L iron)	90 µg/L ±28 (n = 5)	35 µg/L (at midpoint of process)
Ferrous/ferric (~1200 mg/L ferrous/1200 mg/L ferric iron)	563 µg/L ±280 (n = 5)	409 µg/L
Recycle Sludge (~2340 to 13,290 mg/L iron)	387 µg/L ±58 (n = 12)	77 µg/L
	Catalyzed Cementation Results	
Treatment Condition	Mean Selenium Effluent Concentration (µg/L) ±Standard Deviation (n = sample size)	Minimum Selenium Effluent Concentration (µg/L)
Catalyzed Cementation	834 µg/L ±204 (n = 42)	193 µg/L
Catalyzed Cementation with Increased Oxidation/Decreased pH in the reactor tank	35 µg/L (n = 2)	26 µg/L
Additional Testing of Catalyzed Cementation at MSE	3 µg/L[1] ±4.4 (n = 5)	<1 µg/L
	BSeR™ Process Results	
Residence Time	Mean Selenium Effluent Concentation (µg/L)[2] ±Standard Deviation (n = sample size)	Minimum Selenium Effluent Concentration (µg/L)
12 hrs (Series 1)	8.8 µg/L ±10.2 (n = 17)	<2 µg/L
11 hr (Series 2)	4.9 µg/l ±4.9 (n = 16)	<2 µg/L
8 hr (Series 3)	<2 µg/L ±2.6 (n = 12)	<2 µg/L

5.5 hr (Series 2)	<2 µg/L ±2.1 (n = 26)	<2 µg/L

[1] Nondetects were substituted with 50% of detection limit (0.5 µg/L).

[2] Nondetects were substituted with 50% of detection limit (1 µg/L).

Table 15: Economic analysis of demonstrated technologies for selenium removal (MSE Technology Application Iinc., Activity III, Project 20, 2001)

Cost	**Ferrihydrite Adsorption**	**Catalysed Cementation**	**BSeR^{TM} Process**
Capital	$1,026,835 (includes system design, demolition, building modifications, equipment purchase and installation, construction, system start-up, commissioning, and project closeout)	$1,083,285 (includes additional research and development work, system design, demolition, building modifications, equipment purchase and installation, construction, system start-up, commissioning, and project closeout)	$603,999 (includes biofim support material, inoculum, system design, building modifications, equipment purchase and installation, construction, commissioning, and project closeout)
Annual Operating and Maintenance Cost	$2,084,559 (includes reagent costs, manpower, maintenance, and power for equipment use)	$1,165,358 (includes reagent costs, manpower, maintenance, and power for equipment use)	$135,029 (includes nutrient costs, manpower, maintenance, and power for equipment use)
Net Present Value of Annual Operating and Maintenance Costs	$16,992,127	$9,499,323	$1,100,682
Total Net Present Value	$18,017,962	$10,582,608	$1,704,681
Net Present Value of $/1,000 gallons treated	$13.90	$8.17	$1.32

6.0 SUMMARY OF THE COSTS FOR THE PIT LAKE WATER TREATMENT PROCESSES

The costs of the pit lake water treatment processes are summarised in Table 16. Where possible, the costs are standardised for treatment of one cubic metre of effluent and all costs are reported in Canadian dollars.

Table 16: Summary of the costs of various treatment processes

Process	Amendment	Process	Volume	Cost (Cdn $)
Island Copper Mine pit lake	fertiliser	bioremediation	3,760,000 m^3/yr	$0.026/$m^3$ ($100,000/yr)
Sweetwater pit lake	sugar, alcohol phosphate	bioremediation		$17.17/$m^3$
Berkeley pit lake	lime	Precipitation/ adsorption	28,300 m^3/day	Plant: $15.7-$28.8 million Operational: $2.6 million/yr
Berkeley pit lake KAD process		adsorption	11,355 m^3/day	$264/$m^3$
Berkeley pit lake arsenatephosphateapetite		precipitation	1,635 m^3/day with 500 µg/L As	$0.26/$m^3$
Ferrihydrate Se removal		precipitation		$4.75/$m^3$
Berkeley pit lake catalysed cementation of selenium		cementation		$2.83/$m^3$
Berkeley pit lake BSeR selenium removal		bioremediation		$0.46/$m^3$
CANMET	lime	precipitation		$0.012/$m^3$
CANMET	limestone	precipitation		$0.003/$m^3$
CANMET estimate				$0.20/$m^3$

7.0 PHYSICAL METHODS OF REMEDIATION OF PIT LAKES

Stottmeister *et al.,* (1999) described several physical *in-situ* approaches for producing stratification as a means of mitigation for pit lakes. These ideas were collected from the literature, field experience and laboratory experiments (Fichtner 1983, Benndorf 1994, and McCleary 1994). The methods for developing stratification are as follows:

1. Changing a polymictic (multiple mixing periods per year) lake into a stable stratified dimictic (two mixing periods per year) lake by floating and submerged barriers to reduce mixing though wind action.
2. Revegetation of the mine site to increase evapotranspiration and reduce oxygen transportation through the subsoil.

3. Rapid flooding of the pit mine with surface water.
4. Increase organic matter in the sediment layer to consume oxygen and create anoxic conditions for bacterially-mediated H_2S production.
5. Install upgrowth reactors on the bottom of the pit lake to produce anoxic conditions.

7.1 Pit Flooding With Clean River Water

Fast flooding of open pit mines with neutral water can reduce the rate of acidification and stabilise the pit walls. Once the rock is covered with water and anoxic conditions develop in the lake, the rate of oxidation of the sulphide minerals in the pit rock wall is reduced. Fast flooding was used in reducing acid generation from pit rock walls and backfilled wastes at the Enterprise pit in northern Australia, Island Copper Mine pit lake and Lake Goitsche in Germany (Sinclair and Fawcett 1994; Poling et al., 2003; Schultze *et al.*, 2003; Boehrer *et al.*, 2003).

Boehrer *et al.,* (2003) and Schultze *et al.,* (2003) reported the progression of flooding and neutralisation of Lake Goitsche, a pit lake near Bitterfeld, Germany. after flooding with river water. River water started mixing with residual acid waters at the base of the open pit lignite mine in 1999 and reached maximum depth of 48 m in 2000. Lake Goitsche consists of three connected sub-basins with a surface area of 13.3 km^3. The first sub-basin filled until it crested the sill separating it and the adjacent sub-basin. The two sub-basin filled simultaneously until the water level crested the separation to the third sub-basin. When the water level was equal in the three sub-basins the water flow was unidirectional and further water exchanges between basins was very limited. This allowed different evolution of pH in the separate basins. The annual temperature stratification in the basins determined the path of the introduced river water could take. This system controlled the sequence and the time when neutralisation happened in the water layers in each sub-basin. The authors reported that a neutral top layer of water was overlying an acidic bottom layer for the period of thermal stratification. This is similar to the situation in the Island Copper pit lake, except the stratification results from a salinity gradient. The water in Lake Goitsche has been completely neutralised except for a small volume of water in the deepest locations of the lake. These anoxic waters show stable stratification with a different chemistry compared to the water in the rest of the lake. Groundwater flow could be contributing dissolved metal levels to the waters in the deepest parts of the lake.

A somewhat similar situation was artificially created in Lake Fuchskuhle, a small moor lake in eastern Germany. The lake was divided into four basins with plastic sheeting to establish stratification in each basin. Each basin essentially became a small lake with a high ratio of surface area to depth. The basins became stratified with anoxic deep waters and sediments The pH of the waters in the basins changed to nearly neutral and bio-productivity increased (Babenzien, 1996, reported by Stottmeister, 1999).

8.0 CONCLUSIONS OF THE LITERATURE REVIEW

Development of treatment strategies for water contained in pit lakes is dependant on site specific needs. Each pit lake will have unique, geographical, meteorological, and environmental characteristic, and the chemistry in each pit lake is different. A mitigation strategy that is proven successful at one pit lake may not necessarily achieve the same water quality results at a different pit lake.

The waters in the Faro and Vangorda pit lakes have elevated concentrations of Mg, Mn and Zn and will need to be treated before being discharged into the receiving environment. One closure option identified for the two pit lakes is to reduce the sources of metal loading within the area of the pit lakes and redirect the flow of nearby creeks to the lakes. The water leaving the lakes would then be discharged into the original creek channels. In general, this strategy should improve the water quality as the contaminated waters in the pit lakes are replaced with clean water. The unknown factor in this strategy is the metal loading carried into the lakes from the rock walls and watershed in the vicinity of the pit lakes. Seasonal variation in the water flow from the creek into the lake can affect the metal concentration in the pit lakes. Active treatment could still be required to achieve discharge water quality.

Acid generation, at Faro and Vangorda mine site, will continue for many years. A treatment process for the waters from the two pit lakes must ensure long-term environmental protection and minimize the economic risks. An effective mitigation strategy should be designed to accommodate possible changes in site conditions or weather events that could make the system less efficient.

Chemical treatment methods are technically feasible but depending the volume of water that has to be treated from a pit lake, the cost may be high. Recent demonstrations of innovative technologies at Berkeley pit lake indicate that some treatment processes did not meet all the parameters specified in the objective for water quality. Based on the results of the demonstrations, the US-EPA record of decision was to use lime treatment to meet the objectives for water discharge from the pit lake.

Lime treatment is a proven technology that can remove dissolved metals, indefinitely, from large volumes of water and meet long-term environmental protection at these sites. Potential drawbacks for lime treatment is that it may not meet permit water discharge to the environment where sulphate and/or hardness and discharge pH are limiting factors.

Biological treatment methods are a promising new technology for *in-situ* mitigation of pit lakes and the implementation and operational costs calculated from *in-situ* tests is considerably less compared to chemical treatment. However, adsorption of metals by phytoplankton and bacterial-mediated precipitation of metal as sulphides have not been demonstrated to be reliable *in-situ* treatment methods over the long term. Biological processes do not cope well with sudden changes to the system such as an increase in metal loading, water flow or loss of anoxic conditions in deep waters of the pit lake. Biological systems require a long time to re-establish equilibrium in the system.

Island Copper Mine has successfully used *in-situ* biogeochemical processes in the pit lake to meet permit water quality in the top water layer. The pit lake system is still in flux (Fisher and Lawrence 2002) and the long-term performance of biogeochemical processes is unknown. This pit lake is also unique because seawater was used to quickly fill the open pit. The stability of the stratified saline seawater and the biological processes could be very different compared to a pit lake that is filled with mine drainage.

9.0 INFORMATION AND DESIGN REQUIREMENTS

Treatment methods to reduce metals in drainage include a diverse group of processes. The treatment technology with the longest track record is active chemical treatment. Throughout the world there are numerous examples of chemical treatment plants that have operated successfully for decades or more. The speed, robustness and reliability of chemical treatment has made it the technology of choice where the contaminant loads are high or there is no opportunity for experimentation. In British Columbia there are twelve sites with acidic drainage where long-term lime treatment occurs now or is planned and three sites have or are using ferric sulphate to treat Mo and Sb in neutral pH drainage.

However, while treatment with lime and ferric sulphate can be very effective, there are a number of drawbacks. These include the potentially high costs and challenges with sludge disposal. Consequently, a number of mines are considering biological and other less-expensive treatment measures. Examples of the kinds of situations where alternative treatment methods are being considered include treating small ephemeral, acidic pH flows with elevated metals (e.g., Duthie Mine), drainage in flooded mine workings (e.g., Equity Silver and Huckleberry) and low S wastes, where a relatively small decrease in drainage metal levels is required to avoid impacts (e.g., Highland Valley Copper). As a result of all the interest, there has been an increase in field-scale test-work and an increasing number of full-scale applications (Harrington, 2002).

Site conditions play a major role in determining best management practices, and each site and procedure must be assessed on its own merits. However, there are also a number of generic information and design requirements. These include the following.

Location, Quality and Quantity of Contaminated Drainage: Need to know volumes and loadings and for pit lakes, spatial changes within the water column. Need to also know the input and discharge locations, flow rate (retention time), drainage chemistry, pO_2 and interaction among key environmental parameters (e.g., solubility constraints). Must predict both seasonal and long-term variability. These items will require an understanding of future changes in site hydrology and geochemistry. The pit lakes at the Equity mine illustrate the potential stratification of contaminants and changes among key environmental parameters with depth. At Equity, the highest dissolved metal levels exist in a thin layer at the surface. Equity avoids treating pit water by discharging the low metal drainage below this layer and above the depth filled with sludge.

Drainage Collection/Storage System: Effective drainage collection is a critical component of a treatment system. Drainage storage will depend on the hydrology, the treatment rate, resulting effluent quality and permissible discharge. If the treatment rate is limited, pre-treatment storage may be required to handle high flows. Post-treatment storage may be required for adequate dilution during discharge. A collection issue that may arise for pit lakes is the fate of contaminated drainage losses to groundwater if only the surface layer or overflow is remediated.

Treatment Process: A treatment proposal should describe the proposed treatment process (chemical and/or biological reactions) and required conditions, such as pO_2, pH and redox, outline facilities, resource and management needs, predict post-treatment effluent quality and volumes over the range of predicted flows and input water quality, and indicate procedures for verifying the predicted performance (science-based). The treatment process must be reliable and sustainable. Site-specific field-scale testing and detailed, operational monitoring are likely to be required for technologies that are more complex, lack previous use, or where there is no back-up. An important component of treatment reliability is the degree of operator vigilance and control. Monitoring should be adequate to guide management. This includes monitoring key properties and processes, and the quality and quantity of the treated drainage and the resulting effluent.

Notably, while treatment costs at Island Copper are only approximately $100,000/year, much more than that is spent annually on studies and monitoring to inform management and ascertain long-term sustainability. Studies done on other types of biological systems point to the need for sufficient retention time, which for pit-lakes may mean measures to slow the overflow of surface contaminant inputs.

Treated Effluent Discharge: Requirements will depend on the effluent quality, quantity, discharge location(s), and authorized discharge limits and conditions. The latter are the goal posts for treatment and may play a significant role in determining which treatment processes can do the job. Discharge location may have a major impact if it significantly effects dilution and attenuation.

Disposal of Secondary Waste Products: The proponent needs to predict the quality and quantity of any secondary wastes. Based on this they should devise a disposal plan that provides adequate storage space, physical and geochemical stability, and considers future hydrological and ecological developments. Ongoing costs will include those for disposal and for monitoring the composition and volume of both the waste and drainage from the disposal site.

Identify and Minimize Risk: Measures to minimize the likelihood of failure include an ability to perform under extreme climatic conditions, comprehensive monitoring, frequent maintenance, well prepared contingency measures and having sufficient financial capability. Contingency plans typically include back-up power and pumps, spare parts, amendments in the event access is cut-off and excess storage for contaminated drainage. It is especially important for biological processes that systems are properly sized and protected from overload conditions (Gusek and Wilderman, 2002). Other lessons learned by Gusek and Wilderman (2002) on flow through systems that may be applied to pit-lake treatment are that insurance alkalinity may be required to protect SRB from water quality excursions and the importance of pilot studies and the need to ensure amendments are available is sufficient quantities.

Capital and Operating Costs: Along with effectiveness and reliability, existing and projected future capital and operating costs should determine which treatment strategy is selected. This information may also be used by regulatory bodies to ensure the proponent has the required resources. Predicting future costs will be very difficult without a good understanding of key inputs and for relatively new treatment technologies. Without this information, it will be difficult to compare the cost/benefits of alternate treatment measures. Significant costs that are sometimes overlooked include the costs of monitoring, maintenance and access. Access may be a large cost in remote areas or harsh climates, especially if long-term treatment requires permanent road access.

10.0 RECOMMENDATIONS REGARDING PIT-LAKE TREATMENT AT FARO

Based on the limited information available, three processes of pit-lake treatment appear to be potentially feasible.

1. Lime or some other similar form of chemical treatment to raise the pH.
2. Amendments with sugar and alcohol to create anoxic conditions under which SRB precipitate metals.
3. Nutrient additions as means of creating algae and phytoplankton that remove metals such as Zn when they settle to the bottom.

Key questions (see previous section) with all three systems are:

1. At how high a metal load or flow rates can the system reliably meet permissible discharge concentrations, for how long and at what cost?
2. What is required in terms of process control, waste disposal, equipment, personnel, monitoring and maintenance, and discharge?

Lime treatment has been used successfully in the past for pit lake treatment at Samatosum in B.C. and is still used for lake treatment at the Levack mining complex in Ontario. The relatively low acidity in pit water suggests that lime requirements will be relatively small. Depending on whether there is a local source the major cost may be transportation. Questions and concerns that should be addressed with potential lime use include:

- the lack of Fe to co-precipitate trace metals such as Zn,
- whether treatment should occur in the pit or only to the overflow,
- whether additional measures will be required to lower the resulting pH prior to discharge and
- how to prevent discharge of the resulting precipitates.

According to a recent paper by Harrington at the recent BC ARD workshop (2002), treatment of pit lakes with sugar, alcohol and proprietary amendments has been very successful in creating anoxic conditions under which SRB precipitate metals. This presentation outlined work at a number of sites including Brewery Creek in the Yukon. Notably pit treatment avoids the plugging problems, a major limitation with flow through SRB systems (Gusek and Wildeman, 2003 and Tsukamoto & Miller, 2003). A major limitation for SRB reactors is the input of oxygen in drainage or from surface processes such as wind action. Oxygen inputs with the drainage are why SRBs are unable to treat ARD injected into the pit lake at Island Copper pit lake. A back of the envelope calculation of potential reagent costs based on the data provided is as follows:

Assumptions: Metals to be removed by sulphide precipitation are 20 mg/L Mn, 70 mg/L Zn, 1:1 molar ratio addition of ethylene glycol to sulphide and 10 ppm dissolved O_2 must be removed at 0.38 USD/ lb (Chemical Market Reporter, October 13, 2003)

- Theoretical ethylene glycol required is: 95 mg/L
- Using a basis of 1000 L or 1 m^3 this would be 95 g/m^3

Final cost for treatment would be approximately \$0.104 /m^3
As a safety factor, this could be multiplied by 2, which would make the cost 0.208 \$/m^3.

Lime or limestone cost to treat 500,000 m^3 of effluent is as follows:

- Hydrated lime at 96% purity and 90% efficiency, the cost for treatment would be \$6,400
- Limestone at 90% purity and 60% efficiency, the cost for treatment would be \$1,650.

Questions and concerns that should be addressed regarding this strategy include:

- compatibility with water management limitations and requirements,
- oxygen inputs and the quantity and frequency and method of amendment application, and the resulting costs,
- how to sustain anoxic conditions, and deal with potential oxygen inputs from events such as rock fall, high runoff and high winds,
- measures required to address site-specific conditions, and
- how to address the issue of proprietary amendments (e.g., nutrients).

Nutrient additions as means of stimulating the growth of phytoplanktonic algae, have been successful at least in the short-term both at Island Copper Mine and at Landusky pit lakes (Adams, 2002). Successful results were also obtained from pilot-scale tests at Equity Silver mine. Equity has a cold climate that although not as harsh may be comparable to Faro. The treatment at Landusky also involved inoculation and chemical treatment. Questions and concerns that should be addressed regarding this strategy include:

- compatibility with water management limitations and requirements,
- the quantity and frequency and method of amendment application, and the resulting costs,
- measures required to address site-specific conditions, such as how to deal with seasonal constraints on biological activity, and
- how to address the issue of proprietary amendments (e.g., nutrients).

Temperature will be a concern at Faro, both from its potential impact on biological reactions and management. One way to limit this will be to treat and pump, treating the water during the summer and then pumping the pit lake down sufficiently so it can hold contaminated drainage during the rest of the year and there is no discharge until after the next treatment occurs. This appears to be what was done at Brewery Creek. A potential downside is if pumping down the pit results in significant contaminant input from newly exposed mine walls.

Water management is a potential issue with all three treatment measures. The Brenda mine, which treats Mo in its pit lake water in a downstream treatment plant, draws down the level of its pit lake so that a large rock fall will not cause contaminated discharge to the environment. Another water management issue is the best place for dilution to occur. The Huckleberry mine is presently evaluating whether to add the alkalinity and dilution provided by ‘clean’ drainage into the pit lake or add it after the drainage leaves the pit lake. If diluted water still requires treatment, the extra volume can increase costs and cause other difficulties. Dilution may also cause dissolution of precipitated metals if they are at the solubility threshold, maybe reducing concentrations but increasing loadings. The ability to control uncontaminated drainage could therefore be quite important. Diversion of uncontaminated drainage away from unflooded benches will reduce contaminant additions to the pit lake.

Obviously more information is required regarding contaminant loadings, flow, retention time and oxygen inputs for a proper evaluation. Notably the data provided on pit lake chemistry did not include As, Mo, Se and Sb concentrations. Ferric sulphate is usually used to treat drainage where these elements are present in elevated concentrations.

The assessment of treatment alternatives for the pit-lakes should be expanded to consider the benefits of other complementary mitigation measures, such as source controls, that might significantly reduce treatment costs (maybe even the need) and most cost-effectively meets long-term discharge requirements. Much of contamination from upper unflooded portions of pits often comes from talus that accumulates overtime on benches rather than the walls themselves. One effective control measure for this contaminated drainage source is to reduce the number of benches so that detached rock falls into the pit lake. So-called passive treatment systems such as limestone drains may be used to reduce contaminant input from small seeps. As for treatment of the pit-lakes itself, the evaluation and design of associated measures should consider factors such capacity, reliability, longevity, monitoring and maintenance.

The evaluation of treatment options for the pit-lakes also needs to consider the mitigation requirements for the site as a whole. At Equity Silver, despite input of ARD from walls above the Main Zone pit lake, pit water is dischargeable because of the alkalinity added with treatment sludge created from the lime treatment of waste rock ARD. At Faro, it may be cost-effective to treat pit-lake water with drainage from the waste rock or tailings.

Lastly in an ideal world, a single treatment would result in the re-establishment of biota that would naturally remove contaminants to the required concentration. While this may be the final goal, in reality at most mines, responsible management includes regular inputs, replacement, monitoring and maintenance. The goal is how to most cost-effectively achieve conduct of this work. It is also important to note that there are no cook books. Best management consists of the tools for developing the required site-specific understanding and mitigation plans.

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12.0 APPENDIX

Results of the database search for literature on pit lakes.