

## Final Report

# Water Quality in Open Pit Precious Metal Mines

Margaret Saunders Macdonald  
Glenn C. Miller  
W. Berry Lyons

Department of Environmental and Resource Sciences  
University of Nevada, Reno  
Reno, NV 89557

December 1994

Supported by a Grant from the U.S. Environmental Protection Agency

(Grant #R820245)

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## **Abstract**

There are many open pit precious metal mines in Nevada that will be filled with water after mining is completed. This study was conducted to document the current level of understanding of issues concerning pit water quality, and to determine where additional research is needed. A computer-aided literature search was conducted using the keywords pit water, open pit, groundwater, mining, arsenic, geochemistry, wallrock, evapoconcentration, hydrothermal, acid mine drainage and various combinations of these words. There is a wealth of information available about lakes. However, pit lakes are a new type of lake, with unique geometry and geochemistry giving them characteristics unlike those of natural lakes. Factors that contribute to pit water quality include flow of groundwater, water-wallrock reactions, pH, trace element concentrations (especially arsenic), evapoconcentration and hydrothermal activity. Pit lakes exist from the mining of minerals such as phosphate, uranium, coal, copper, silver and gold. None of these lakes have been studied in extreme detail, but limited studies contribute knowledge to the understanding of water quality in open pit mines. The variability in the contributions of the factors that affect pit water quality and the unique geology in each mine necessitate the study of each pit lake or future pit lake on an individual basis, and make prediction of water quality difficult.

## **Disclaimer**

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In the electronic version of this document, the figures and tables are included as a separate file.

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## **Introduction**

The development of technology that allows heap leaching of disseminated gold ores by cyanide solutions led to the profitable mining of gold ores with average grade as low as .02 ounces per ton (NBMG, 1989). This means that, on average, 50 tons of rock are removed to produce each ounce of gold. With over 5 million troy ounces of gold produced in Nevada each year (NBMG, 1991), a large amount of rock is removed from open pit mines. Many of the mines are now dewatering to retrieve ore below the water table. As mining goes deeper sulfide ore bodies are encountered rather than the oxide ore bodies found closer to the surface. When dewatering stops and the water table in the pits returns to its original level, the mines will contain pit lakes. Figure 1 shows open pit mines in Nevada that are currently water-filled, are now dewatering, or will be dewatering in the future. These lakes will exist long after mining ends, causing environmental concern for centuries.

Because open pit precious metal mining below the water table is a relatively new phenomenon, there is a lack of information regarding water quality in these pits. The only discussion of pit water geochemistry is that of Davis and Ashenberg (1987) who studied the Berkeley Pit lake in Butte, Montana. The objectives of this research paper were to document the current level of understanding of issues concerning pit water quality, and to determine where additional research is needed.

A literature search was conducted using Silver Platter software and the Georef and Selected Water Resources Abstracts databases. The keywords used for the search were pit water, open pit, groundwater, mining, arsenic, geochemistry, wallrock, evapoconcentration, hydrothermal, acid mine drainage and various combinations of these words. All information for each citation was searched (title, abstract, keywords, etc.), and all years in the databases were included.

There is a wealth of information available about lakes (Hutchinson, 1957; Wetzel, 1983). All of the natural lake types have been studied and classified according to

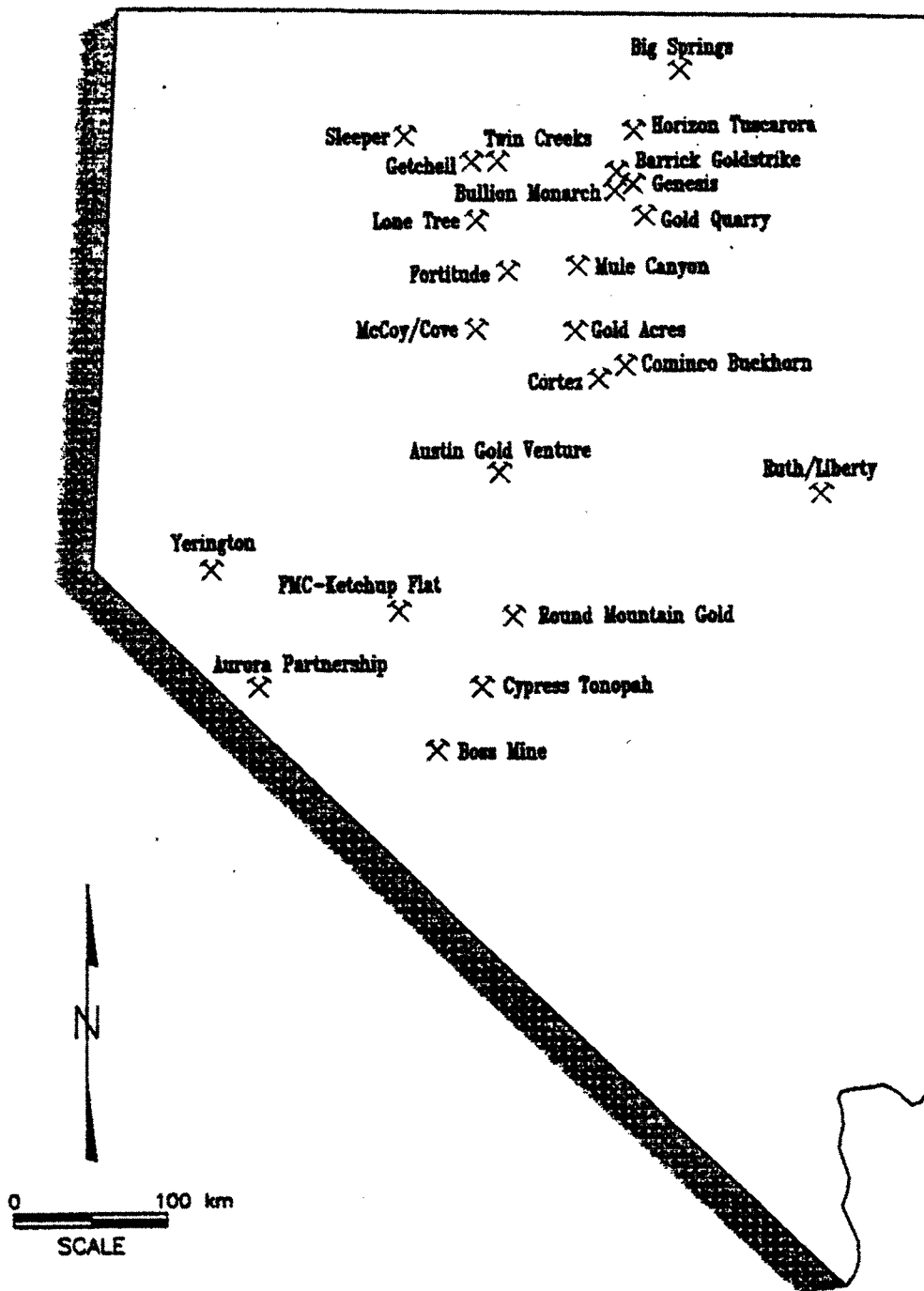


Figure 1 Location map for Nevada open pit mines requiring dewatering

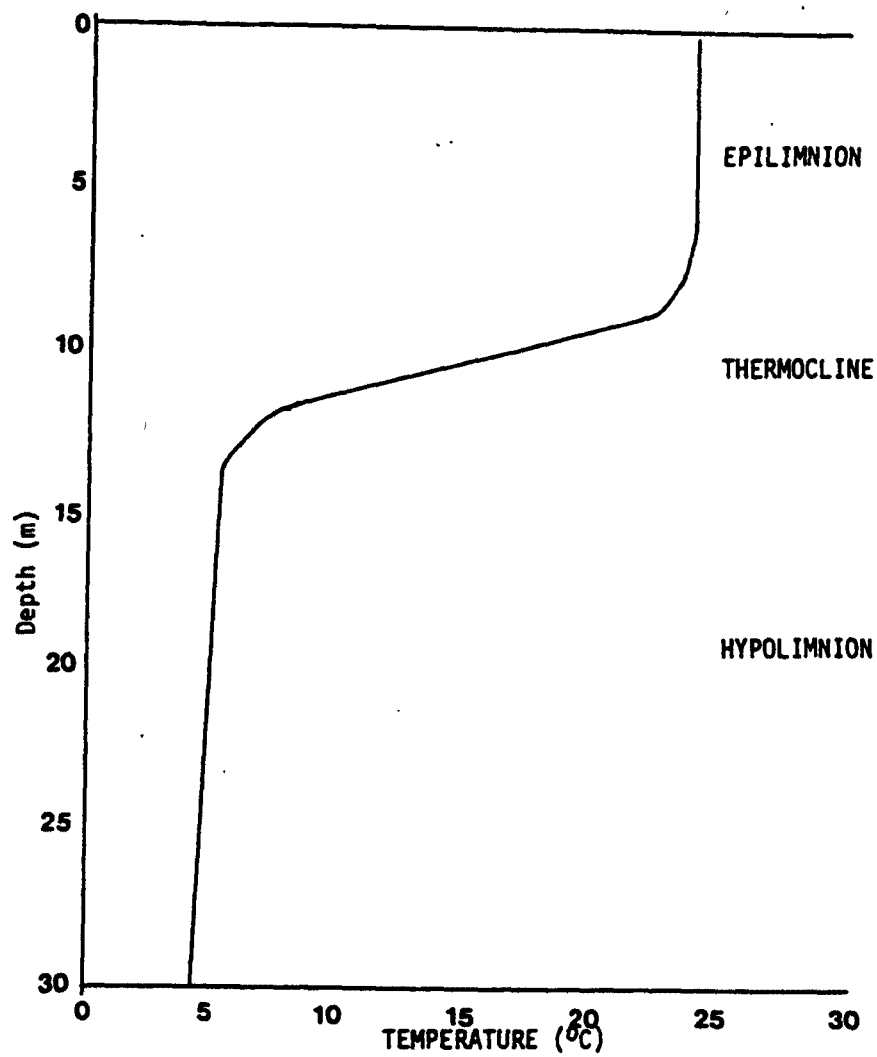
circulation and stratification patterns. Biologists know in detail the types of aquatic biota that are found in natural lakes. Man-made reservoirs have been studied and found to have fundamentally different characteristics from natural lakes, but those characteristics are easy to understand since the dynamics of natural lakes are known and both natural lakes and man-made reservoirs are usually relatively shallow (less than 20 m deep) (Wetzel, 1983). Mining pit lakes, however, are a new type of man-made water body due to their extreme depth (often over 300 m deep). Their surface area to depth ratio is small, making them most similar in shape to natural volcanic crater lakes such as Crater Lake in Oregon. Newly exposed wallrock in pit lakes, with the mineralogy associated with ore deposits, can contribute chemical constituents in higher concentrations than are found in most natural volcanic lakes.

What follows is a review of what is known about the behavior of natural lakes (Wetzel, 1983), which will give an idea of the possible behavior of pit lakes. However, definitive answers will not be available until the pits are filled with water and limnological studies are completed on them.

Thermal stratification occurs in a lake when the surface waters are heated by solar radiation more rapidly than the heat is dissipated by mixing. The temperature profile of a typical stratified lake has three zones (Figure 2): warm water in the epilimnion, cold water in the hypolimnion, and a zone of rapid temperature change with depth called the thermocline. The maximum density of water occurs at 4°C so this is a very stable profile, with the densest water at the bottom.

Lakes can undergo seasonal changes in stratification (Figure 3). The most stable stratification profile, with maximum temperature difference between the epilimnion and the hypolimnion, occurs in summer. Oxygen in the hypolimnion varies seasonally and can go to zero under stratified conditions as shown in Figure 3, depending on whether oxygen-consuming materials are present. An oxygen deficit in the hypolimnion is usually due to oxidation of organic matter but can also be caused by chemical oxidation (Hutchinson, 1957). As the surface waters cool in the fall, the temperature profile becomes less and less stable.





Typical thermal stratification of a lake.

After Wetzel, 1983  
Limnology

Figure 2

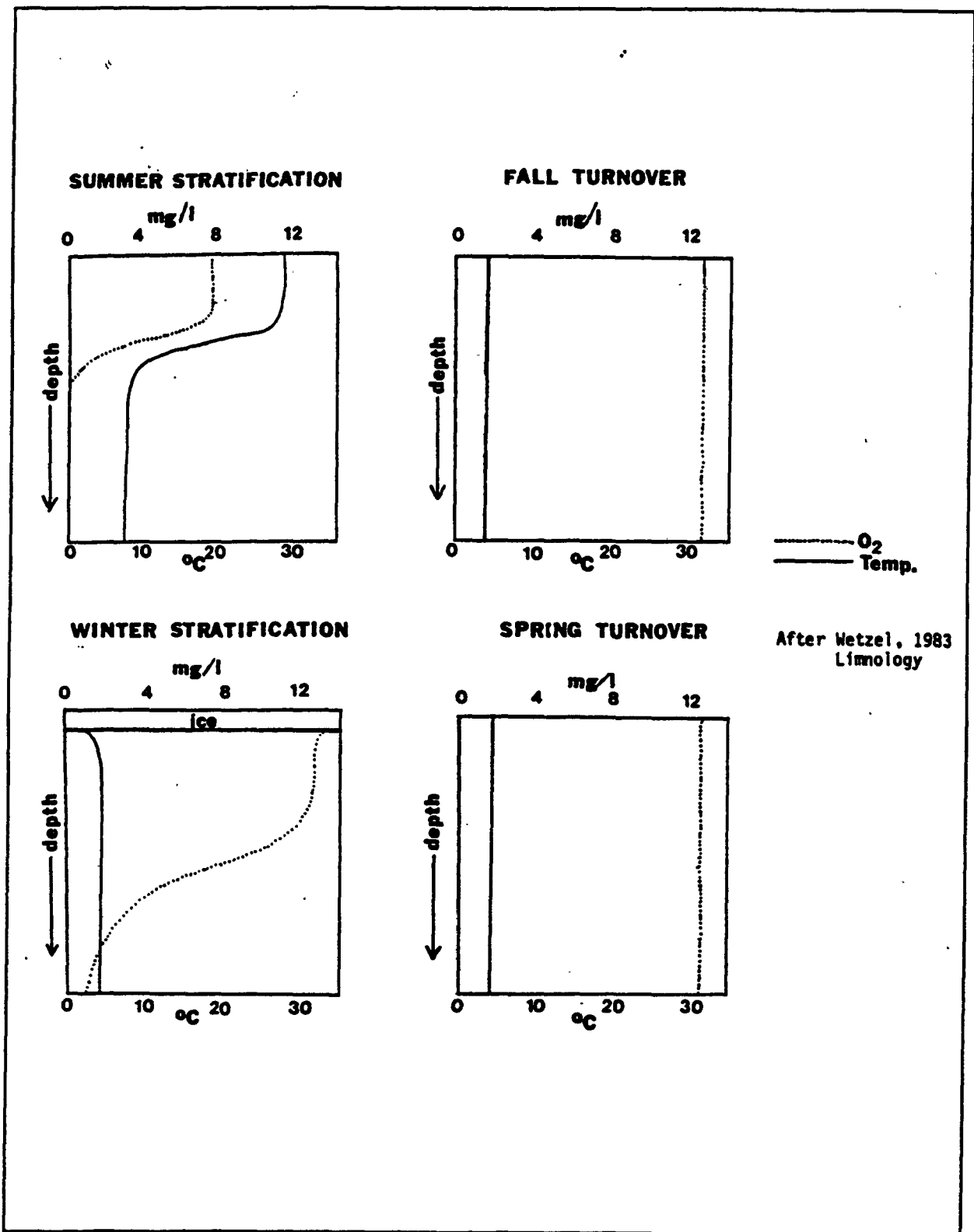


Figure 3

Turnover may occur, generally on a windy day when the water in the epilimnion is nearly the same temperature as the water in the hypolimnion. The entire lake will mix and oxygen content will be high throughout (Hutchinson, 1957). In winter, ice may form as the surface water cools, and the lake is once again stratified. Then, as the surface warms in the spring, the ice melts and the profile again becomes unstable due to nearly isothermal conditions. If the lake receives sufficient energy from wind, again turnover and mixing occur, and again oxygen contents are high throughout the profile.

The water in the Berkeley Pit in Butte, Montana did not freeze in the winters of 1988-89 or 1989-90, but did freeze in earlier years (Huang and Tahija, 1990). In the years that the Berkeley Pit froze, immediately after the ice melted in the spring the surface water of the pit was a blue-green color. By midsummer, it was brown in the upper 4 m and blue-green below 4 m (Davis and Ashenberg, 1989). The blue-green color represents reducing conditions (a Cu(II) mineral). As the surface waters became oxygenated after the ice cover was gone, iron oxyhydroxide precipitated giving the brown color. Ice may play a role in lowering surface metal concentrations due to exclusion of soluble ions as ice forms. The unfrozen solution below the ice is then enriched in soluble ions and may sink due to higher density (Huang and Tahija, 1990).

A high concentration of total dissolved solids at the bottom of a lake may cause a dense layer of water that does not participate in mixing. This dense layer is called the monimolimnion and the mixing layers above are called the mixolimnion. The steep salinity/density gradient between the layers is called the chemocline. Lakes with this density profile are called meromictic. Cryogenic meromixis occurs from "freezing out" of salts from ice, which then precipitate and sink to deep water, as mentioned above. Crenogenic meromixis is caused by submerged saline springs, and biogenic meromixis occurs from salts released during the decomposition of organic matter. In a meromictic lake the mixolimnion goes through the same circulation and stratification patterns as non-meromictic lakes.

There are many types of stratification patterns that occur in lakes which depend on climate, morphometry and chemistry. Lakes which undergo complete circulation in the spring and fall and are thermally stratified in the summer and winter are called dimictic lakes.

Lakes in which temperatures never go below 4°C circulate at or above 4°C all winter and stratify in the summer. These are the most common patterns among temperate lakes.

Whether or not pit lakes will mix is difficult to predict. Density differences due to concentration of total dissolved solids, due to input from hydrothermal springs, due to climate, and due to pit lake morphometry all contribute to the stability or lack of stability of a lake's density profile. More studies of the limnology of existing pit lakes are needed. Measurements should be taken at different times of the year to provide information about seasonal changes.

## **Factors Contributing to Pit Water Quality**

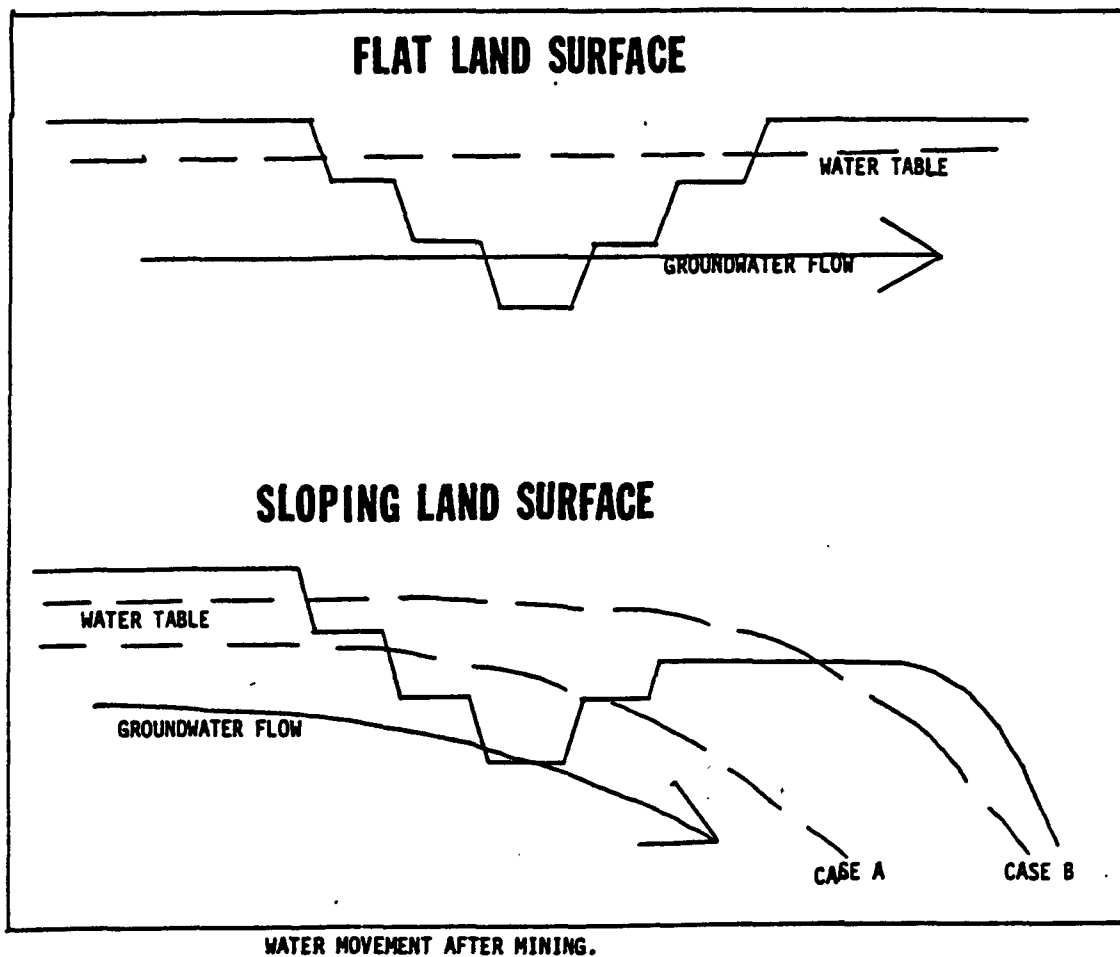
Many factors will contribute to the quality of water in the pits. The pit water may be acidic or alkaline, arsenic concentrations may be high or low, other trace metals may be high or low and evapoconcentration may affect their concentrations. Groundwater flow around the pit is important and hydrothermal water may or may not enter the system. Open pit mining significantly increases the rock surface area that is exposed to water and water-rock reactions are a primary consideration in prediction of water quality. In this section each of these factors will be examined.

### **Groundwater flow**

Groundwater flow is an important consideration in determining pit water quality. Conditions prior to mining must be well known before any predictions can be made about pit water. The original water table elevation will generally be the same as the ultimate water level in the pit and thus determines, in part, the depth of the pit lake.

Figure 4 shows groundwater flow around an open pit after mining. Groundwater flow would be expected to flow into and out of the pit through aquifers, but not everywhere as implied in the generalized Figure 4. It is important to know depth and flow rates of all aquifers intersected by the pit. For an open pit mine on a flat land surface, the pit will fill and stabilize near the level of the original water table except for evaporation and additional recharge. Alternatively, on a sloping land surface, the pit may overflow if the original water table elevation is sufficiently high.

Schubert (1980) found that groundwater flow changed direction in coal mine spoil material adjacent to a lake in Illinois, depending on storm precipitation. During high lake levels after a storm the spoil material received recharge from the lake. The lake then receded due to surface discharge and evaporation, and groundwater seeped back into the lake. Groundwater seeps from the White's and Intermediate open cuts at Rum Jungle and



WATER MOVEMENT AFTER MINING.

After Morin, 1990  
Acid Drainage from Mine Walls  
The Main Zone Pit at Equity  
Silver Mines

Figure 4

contributes to the pollution load in an adjacent river (Harries and Ritchie, 1988). At the Nabarlek Uranium Mine, groundwater seeps from the pit when the water level in tailings in the pit is higher than the surrounding groundwater level during the rainy season (McLoughlin et al., 1988).

Dewatering may alter the natural groundwater flow regime. Cones of depression may extend 10 km or more and subsidence may occur within the cone. Up to .5 m subsidence was recorded within the cone of depression of a large (approx. 2.5 km<sup>2</sup>) dewatered open pit coal mine in Poland with maximum drawdown in the pit of 200 m (Wojciechowski and Serewko, 1985).

Direction of groundwater flow can be an important factor in determining the quality of pit water. Wicks et al. (1991) described final-cut coal strip mine lakes which had spoil material on one side, and bedrock on the other side. In these cases, sulfide oxidation occurred only on the spoil material side, not on the bedrock side. In this situation, upgradient lithology determines groundwater quality. Groundwater flowing from the bedrock into the lake would be relatively uncontaminated. If groundwater flowed from the spoil into the lake, lower water quality would be expected.

### **Water-wallrock reactions**

In addition to upgradient lithology, much of the quality of pit water will depend on the interaction between water and the wallrocks in the pit. This interaction will depend on the amount of wallrock exposed to water, how permeable the wallrocks are, how stable the submerged pit slopes are, as well as the redox and chemical conditions in the lake. This section discusses the physical processes that determine the amount of wallrock area exposed to water. The redox and chemical conditions are discussed in later sections.

The area of wallrock surface that will be exposed to water should be computable from the mine plans and knowledge of the original water table elevation. Based on these calculations, one can determine the fixed surface area of wallrock available for reactions. This calculation would, however, be complicated by flow in fractures and joints and by

sloughing of pit walls both above and below the pit water surface, which will provide new mineral surfaces for reaction.

### **Flow in fractures**

Water flow in joints and fractures can greatly increase the rock surface area available for water-rock reactions. The permeability of a system of fractures is much larger than that of a porous medium (Bear, 1979), but the exact amount of permeability is often difficult to determine. In addition to joints and fractures that are naturally present, some will be introduced by blasting during active mining. A fault may have low permeability zones associated with the fault gouge as well as zones of high permeability in the fractured rock. Fault breccia may be more permeable than the gouge. For these reasons, faults can act as groundwater barriers, as groundwater conduits, or both at the same time (Patton and Deere, 1971).

### **Sloughing of wallrocks**

Most of the research on pit slope stability found in the literature pertains to stability in dewatered pits during active mining. The factors affecting pit slope stability are rock structure, including bedding planes, joints and faults; groundwater pressure; earthquakes; weight of a block of rock; hydrostatic forces in joints and fractures; and damage from blasting (Broadbent and Zavodni, 1982; Hoek, 1971; Brown, 1982; Glass, 1982; Piteau and Martin, 1982 and Holmberg and Maki, 1982). Wide fault zones may have large areas of weathered or hydrothermally altered rock and may influence groundwater flow such that pore pressures are excessive within and adjacent to the faults (Patton and Deere, 1971).

An additional factor in determining slope stability is the regional groundwater flow system. Determination of whether the pit is located in a groundwater recharge area or discharge area is important because groundwater discharge areas have a greater likelihood of having excess pore water pressure in the walls and beneath the floor of the mine (Patton and Deere, 1971).



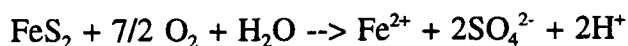
The slope stability factors discussed above apply to wall rock above the water surface in water-filled open pits. No studies were found on the stability of submerged mine walls, although Davis and Ashenberg (1989) indicated that the submerged walls in the Berkeley Pit at Butte, Montana are unstable. Sloughing has filled in 38 m of the pit lake. The Ruth (Nevada) Pit bottom is currently 37 m higher than the original pit bottom (Woodward-Clyde Consultants, 1992) presumably due to sloughing of wall rocks. Additional studies are needed to determine how much sloughing is actually occurring in existing pit water lakes, and how this sloughing affects the water-rock reactions that will take place in the lakes.

### **Acid versus Alkaline Pit Water**

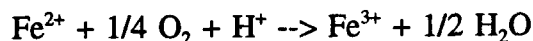
Acid mine drainage (AMD) is considered to be the greatest chemical problem caused by mining (U.C. Berkeley Mining Waste Study Team, 1988). Generation of acids during and after mining results from the oxidation of naturally occurring minerals. Once the AMD process begins, it is difficult to control and often accelerates and is likely to persist for decades or centuries. Several examples exist where AMD began after closure of a mine. After closure, it is difficult to adopt mitigating measures. Acidification of pit water is not expected to be generally observed in precious metal mines. However, the problem may occur in some mines and is of sufficient magnitude that a discussion on the process is given below.

### **The acid forming process**

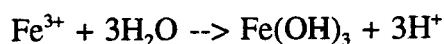
Geology of the wallrocks in the pit and adjacent rocks with which surface water comes into contact are the main influences on the pH of pit water. The amount of acid generated will be determined by the amount of sulfide minerals and/or ferrous iron in the wallrocks and adjacent rocks that are available for oxidation. Oxidation of sulfides forms acid by the following reaction sequence (Steffen Robertson and Kirsten, 1989):



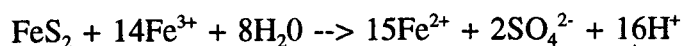
If conditions are sufficiently oxidizing, Fe(II) will oxidize to Fe(III):



At pH values above 2.3 to 3.5, Fe(III) will precipitate as Fe(OH)<sub>3</sub>:



In this reaction sequence, acidification results from the overall production of four equivalents of hydrogen ions. At low pH values (<4), and in the presence or absence of oxygen, pyrite is rapidly oxidized by ferric iron (Nordstrom, 1982; and Nordstrom and Alpers, 1990):



Pyrite oxidation by microbial activity can continue under low or undetectable oxygen concentrations because ferric iron can be used instead of oxygen as an electron acceptor for the bacteria (Nordstrom, 1982). The primary chemical factors which determine the rate of acid generation are pH, temperature (the oxidation rate by oxygen doubles for every 10°C rise, Nordstrom, 1982), oxygen content of the gas phase if saturation is less than 100%, oxygen concentration in the water phase, degree to which a rock is saturated with water, chemical activity of  $\text{Fe}^{3+}$ , surface area of exposed metal sulfide, and chemical activation energy required to initiate acid generation (Steffen Robertson and Kirsten, 1989).

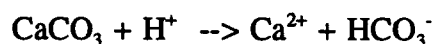
Bacteria such as Thiobacillus ferrooxidans accelerate the rate of ferrous-iron oxidation and of reduced-sulfur oxidation at pH less than 4, and thus accelerate the rate of acid production. As long as pH is maintained above about 4 the rate of acid formation is slow (U.C. Berkeley Mining Waste Study Team, 1988). But below pH of 4, bacteria greatly accelerate the oxidation rate, and once the acceleration occurs it is difficult to reverse. Bacterial growth of T. ferrooxidans is greatest in waters with pH of 3.2, and at temperatures less than 40°C (U.C. Berkeley Mining Waste Study Team, 1988). Above 40°C bacterial oxidation declines rapidly and above 45°C there is little or no bacterial action. Additional factors which determine the bacterial oxidation rate include the biological activation energy, population density of bacteria, rate of population growth, nitrate concentration, carbon dioxide content and concentrations of any bacterial inhibitors. While T. ferrooxidans is the most common sulfur- and iron-oxidizing bacteria there are at least 18 other species of bacteria which can oxidize or reduce iron or sulfur (Steffen Robertson and Kirsten, 1989). In addition to pyrite (iron sulfide), bacteria can accelerate the oxidation rate of sulfides of antimony, gallium, molybdenum, arsenic, copper, cadmium, cobalt, nickel, lead and zinc (Lundgren and Silver, 1980).

Efflorescent iron sulfate minerals (Table 1) can form when sulfides are oxidized but the acid produced is not flushed away and evaporation occurs (Nordstrom and Alpers, 1990).

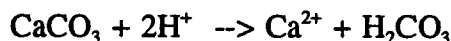
These minerals are highly soluble and "store" acid. Rhomboclase, for example, has acid in its formula:  $(\text{H}_3\text{O})\text{Fe}(\text{SO}_4)_2 \cdot 3\text{H}_2\text{O}$ . These minerals might form in the unsaturated zone, and the first flush of water that comes through would dissolve them, producing a pulse of acid. In this way hundreds of kilograms of acidity could be stored within the walls of a mine (Morth et al., 1972). This stored acidity could be flushed out by normal groundwater recharge during mining or could be washed out during final pit flooding causing mine water to become acidic. This acid water could be mistaken for active acid generation, which may or may not be occurring. Therefore, it is important to determine the rate of acid generation and the movement or lack of movement of water over the oxidation sites (Morin, 1990).

### **Neutralization**

The acid produced by sulfide oxidation may be neutralized if the wallrocks contain sufficient amounts of carbonate minerals:



and



In order for  $\text{CaCO}_3$  to neutralize the acid water generated, it must exceed the equivalent concentration of acid and must also be exposed and available. If amorphous ferric hydroxide precipitates in the area surrounding the reacting carbonate grains, the pH may again decrease due to armoring of the calcite by the amorphous ferric hydroxide (Davis and Runnels, 1987).

Table 1. Iron Sulfate Minerals Identified at Iron Mountain, CA

<u>Mineral</u>	<u>Formula</u>
Melanterite	$\text{Fe}^{\text{II}}\text{SO}_4 \cdot 7\text{H}_2\text{O}$
Ferricopiapite	$\text{Fe}_4^{\text{III}}(\text{SO}_4)_5(\text{OH})_2 \cdot 16\text{H}_2\text{O}$
Copiapite	$\text{Fe}^{\text{II}}\text{Fe}_4^{\text{III}}(\text{SO}_4)_6(\text{OH})_2 \cdot 20\text{H}_2\text{O}$
Roemerite	$\text{Fe}^{\text{II}}\text{Fe}_2^{\text{III}}(\text{SO}_4)_4 \cdot 14\text{H}_2\text{O}$
Kornelite	$\text{Fe}_2^{\text{III}}(\text{SO}_4)_3 \cdot 7\text{H}_2\text{O}$
Coquimbite	$\text{Fe}_2^{\text{III}}(\text{SO}_4)_3 \cdot 9\text{H}_2\text{O}$
Rhomboclase	$(\text{H}_3\text{O})\text{Fe}^{\text{III}}(\text{SO}_4)_2 \cdot 3\text{H}_2\text{O}$
Voltaite	$\text{K}_2\text{Fe}_5^{\text{II}}\text{Fe}_4^{\text{III}}(\text{SO}_4)_{12} \cdot 18\text{H}_2\text{O}$
Additional Sulfates Identified	
Gypsum	$\text{CaSO}_4 \cdot \text{H}_2\text{O}$
Chalcanthite	$\text{CuSO}_4 \cdot 5\text{H}_2\text{O}$
Halotrichite	$\text{Fe}^{\text{II}}\text{Al}_2(\text{SO}_4)_4 \cdot 22\text{H}_2\text{O}$

Nordstrom and Alpers, written  
communication, 1990

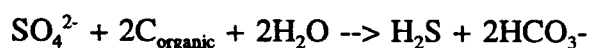
Other minerals consume acid including siderite ( $\text{FeCO}_3$ ), magnesite ( $\text{MgCO}_3$ ), rhodochrosite ( $\text{MnCO}_3$ ), witherite ( $\text{BaCO}_3$ ), ankerite ( $\text{CaFe}(\text{CO}_3)_2$ ), dolomite ( $\text{MgCa}(\text{CO}_3)_2$ ), malachite ( $\text{Cu}_2\text{CO}_3(\text{OH})_2$ ), gibbsite ( $\text{Al}(\text{OH})_3$ ), limonite/goethite ( $\text{FeOOH}$ ), manganite ( $\text{MnOOH}$ ), and brucite ( $\text{Mg}(\text{OH})_2$ ). Each mineral buffers to a different range of pH. Some examples:

<u>Mineral</u>	<u>Buffer pH</u>
Calcite and Aragonite	5.5-6.9
Siderite	5.1-6.0
Malachite	5.1-6.0
Gibbsite	4.3-3.7
Limonite/Goethite	3.0-3.7

(Steffen Robertson and Kirsten, 1989)

Not all of these minerals (e.g. gibbsite and limonite/goethite) can effectively bring the pH up to the neutral range. Silicate minerals also consume acid, but not as rapidly as the carbonate minerals. Inflow of groundwater can also supply acid-neutralizing alkalinity (Morin, 1990).

If conditions are reducing, acidity from acid mine drainage can also be neutralized by microbially mediated sulfate reduction followed by reduced sulfur mineral precipitation because sulfate reduction reverses the acidification process and generates alkalinity. This process also lowers the concentration of iron and sulfate by precipitation of iron sulfides. The equation follows:



The sulfide formed during this reduction process must be stored in a reduced form in the sediments in order for the alkalinity generated to be permanent (Wicks et al., 1991).

Miller and Murray (1988a) outline the stages in the generation of acid mine drainage: Stage 1 - chemical and/or biological oxidation of sulfide minerals which slowly produces acid. The acid may be neutralized by carbonates and result in only a slight decrease in pH, although total dissolved solids may increase. Testing programs that monitor only pH will not identify sulfide oxidation during stage 1. Stage 2 - after the carbonates and other neutralizing materials are consumed, the pH drops and acidophilic bacteria multiply. Stage 3 - when the

pH drops below 3.5, bacterially catalyzed sulfide oxidation becomes effective and the rate of acid generation is rapid.

### **Evaluation of potential for acid production**

The initial characteristics of mine water do not necessarily reflect the long term potential for acid mine drainage, and predicting the acid-forming ability of a rock type is problematic. Also, the time for development of acid conditions can vary from less than one day to more than 50 years (Miller and Murray, 1988a), making it difficult to monitor. However acid producing potential must be evaluated to determine long term water quality. The evaluation is done by static and kinetic tests.

After representative samples of all geologic units in a mine are obtained, static tests are used to define a sample's balance between potentially acid-generating minerals (potential acidity) and acid-neutralizing minerals (neutralization potential). Theoretically, a sample will generate acid only if the potential acidity exceeds the neutralization potential, but as different phases oxidize and/or dissolve the acidity/neutralization balance changes.

The general procedure for conducting static tests is first to measure total sulfur in a sample which gives a determination of potential acidity, and then to determine the neutralization potential. The third step is to calculate the net acid generating potential by subtracting the potential acidity from the neutralization potential. Negative values indicate the potential for net acidity.

There are problems associated with each step of this procedure because the mineralogy and mineral associations in a sample are not considered. The measure of total sulfur may overestimate the acid generating potential if some of the sulfur occurs in non-acid generating minerals, or if some of the sulfur is not exposed and available for reaction in the sample (Miller and Murray, 1988b; Doyle, 1991). The co-existence of pyrite with other sulfides such as chalcopyrite and sphalerite decreases the oxidation rate (Nordstrom, 1982). The net neutralization potential is usually obtained by crushing and blending a sample which is not representative of conditions in the field. In an uncrushed rock, acid may be produced in a

fracture, and may leave the rock via the fracture and never come into contact with neutralizing minerals (Doyle, 1991). The determination of the potential for net acidity is not totally reliable because some mine wastes that contain excess basic material still produced acidic drainage (diPretoro and Rauch, 1988; Erickson and Hedin, 1988).

Despite the limitations of static tests, if they indicate that a sample is potentially acid generating then kinetic tests should be conducted. Lawrence (1990) proposed that if the acid neutralization potential does not exceed the acid generating potential by at least 3:1 there is not a clear margin of safety and kinetic tests should be conducted. Kinetic tests involve weathering samples under laboratory-controlled or on-site conditions to determine rates of acid generation, sulfide oxidation, neutralization and metal depletion.

Steffen Robertson and Kirsten (1989) outline kinetic test procedures as follows. The first step is to determine surface area, mineralogy and total sulfides. Particle size and differences in surface area can account for differences in acid generation rates. The mineralogy of a sample, specifically chemistry and crystal form of the minerals, is important in controlling the rate of acid generation and neutralization (Hammack, 1985; Chander and Briceno, 1988; Hammack et al., 1988). Total metal analyses are important for determining when a metal in a sample may be depleted by leaching.

Problems are also associated with kinetic tests because laboratory tests can not accurately duplicate field conditions and acid generation potential can be determined only by field tests conducted for long time periods. Acid generating rock with significant amounts of both sulfide and carbonate may yield drainage that is neutral for a long period of time before becoming acidic (Steffen Robertson and Kirsten, 1989). Kinetic tests that simulate weathering require at least 6 to 9 weeks to acquire meaningful data, and optimized weathering, using a dilute acid for leaching instead of water, does not show what type of discharge would result from normal weathering conditions (Hammack, 1985). Field kinetic tests up to 20 weeks in length have been recommended in order to exceed the depletion of neutralization potential and not underestimate the acid generation potential (Lapakko, 1990; Lawrence, 1990).

Comparison of pre-mine predictions of acidity or alkalinity from the literature with actual results from the mines at a later time showed that accurate predictions do not necessarily follow from testing programs (Ferguson and Erickson, 1988). A clear and correct prediction can be made only if potentially acid-producing sulfide minerals are greatly abundant or generally lacking relative to acid-consuming carbonate minerals.

### **Mitigation of acid mine drainage**

There is currently no widely accepted technology which controls AMD production at the source without indefinite maintenance (Ziemkiewicz et al., 1990). Steffen Robertson and Kirsten (1989) discuss the three stages of control of acid rock drainage which are control of the acid generation process, control of acid drainage migration, and treatment of acid rock drainage. These methods all may require long term monitoring and maintenance.

Acid generation control involves excluding oxygen or water to inhibit sulfide oxidation, or slowing the acid generation rate by controlling pH or controlling bacterial activity. Oxygen and water can be excluded by use of impermeable covers. pH can be controlled by adding neutralizing materials. Bacterial activity can be temporarily controlled by addition of anionic surfactants such as sodium lauryl sulfate. Inhibition of Thiobacillus ferrooxidans will not reduce acid formation unless the inhibitor is added to groundwater or rainwater infiltration before contact with the pyrite occurs (Kleinmann et al., 1981). Once the acid has left the site of pyrite oxidation, treatment with an inhibitor will not help.

Control of acid rock drainage migration involves diversion of surface water flowing toward the acid source and prevention of groundwater flow into the acid source. These control measures would be difficult, if not impossible, to achieve in open pit mines, although drainage adits are recommended which drain away from the pit as a possible means of long-term isolation of water from sulfide bearing rocks (Steffen Robertson and Kirsten, 1989). In some locations this may be a means of control in wallrocks above the surface of the water in the pit. Surface water can be diverted away from the pit, but precipitation falling at the pit edge will still run down the walls.



Methods for treatment of acid mine drainage may be either active systems requiring continuous operation such as chemical treatment plants or passive systems which operate without human intervention such as wetlands. Steffen Robertson and Kirsten (1989) describe in detail many types of treatment systems, some of which could be used for the pump and treat method of pit water quality remediation.

In summary, in analyzing an open pit mine site for potential acid water, it is important to know the geology of the wallrocks in detail. Until more work on testing methods is done, the advantages and disadvantages of the various static and kinetic tests should be weighed in choosing the test method that will best predict acid producing potential. If acid production is expected, the various ways to prevent it or neutralize it can be analyzed and determination can be made of whether or not it can be controlled before it becomes a problem.

### **Trace Elements**

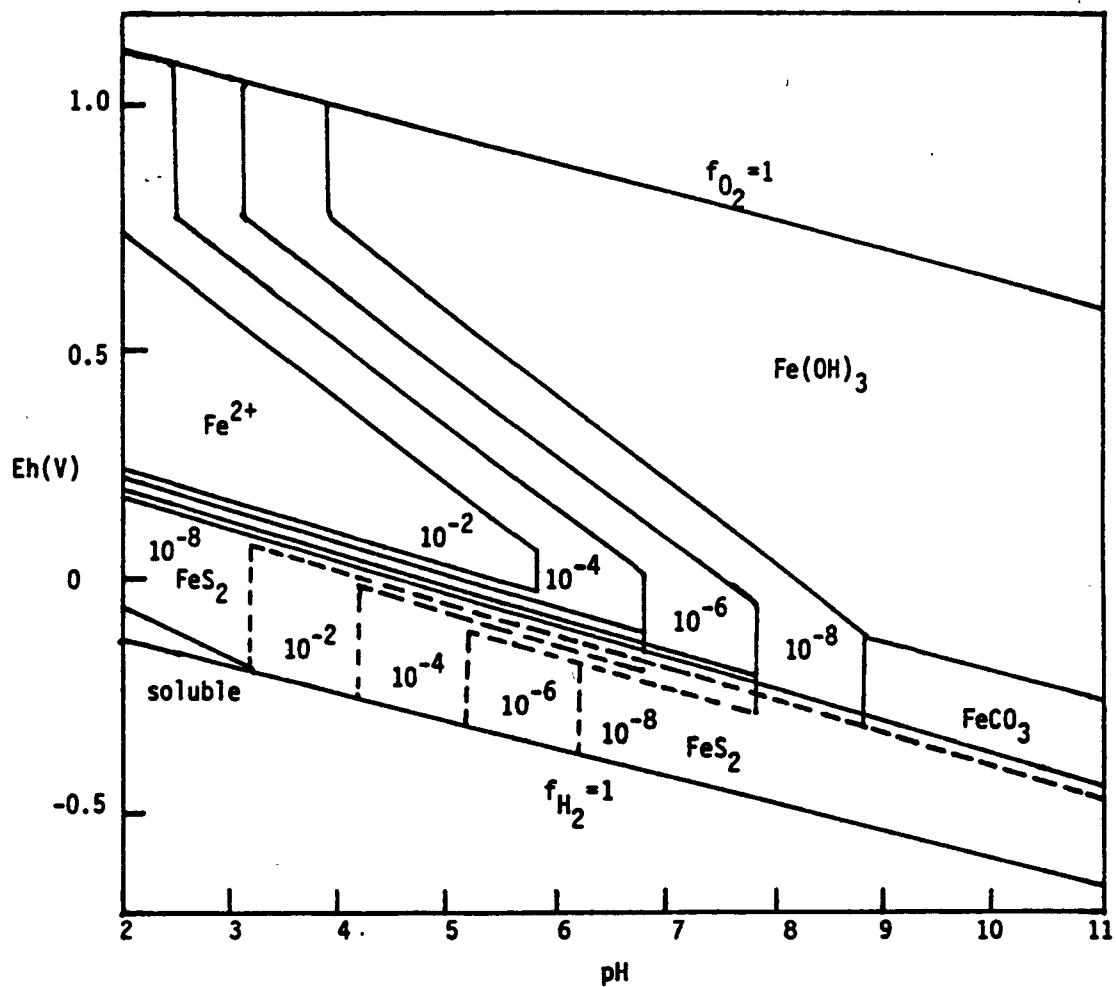
Trace elements are defined as elements that generally occur in waters at concentrations of less than 1 mg/l (Drever, 1988), although in mine waters they are often found at higher concentrations. Trace elements that are of environmental concern are primarily metals and metalloids. Their concentrations are not always easy to predict because in order to calculate the concentration of a dissolved metal in equilibrium with a solid phase, the concentrations of all potential complexing agents and the stability constants of the various possible complexes must be known.

Under oxidizing conditions at high pH concentrations of dissolved trace metals that are positively charged in solution are controlled by adsorption. Tessier et al. (1985) calculated equilibrium constants for the adsorption of Cd, Cu, Ni, Pb and Zn onto iron oxyhydroxides in oxic lake sediments. Adsorption substrates can include clays; organic matter; and iron, manganese, aluminum and silicon hydrous oxides. The relative importance of each type of substrate needs to be determined in order to identify what is controlling adsorption. Relative importance of substrate type will depend on binding ability and abundance. Hydrous manganese and iron oxides both have very high adsorption capacities and affinities for heavy metals. When adsorption by manganese and iron oxides is a control on heavy metal

concentrations in solution, the dissolved concentrations are a function of Eh and pH. Tessier et al. (1985) reported that adsorption of trace metals onto iron oxyhydroxides typically increases from near 0% to near 100% as the pH increases through a narrow critical range of approximately 2 pH units (referred to as the adsorption edge). When the oxides are dissolved by reduction, adsorbed metals are released. In reducing sediments trace element concentrations in pore water are controlled by metal sulfide formation (Moore et al., 1988). As pH increases, the negative surface charge of manganese and iron oxides increases and metals are strongly adsorbed. Optimum adsorption occurs at high pe and pH (Drever, 1988). Based on Eh-pH diagrams (Figure 5), in oxygenated waters dissolved iron concentrations are expected to be low under neutral and alkaline conditions but high under acidic conditions (Drever, 1988).

Decreasing concentrations of aluminum and iron correlate with increasing pH in groundwater near Globe, Arizona, but not well enough to allow accurate prediction of their concentrations based on pH alone (Stollenwerk and Eychaner, 1988). Aluminum solubility appeared to be controlled by microcrystalline gibbsite at pH greater than 4.9. At pH less than 4.9 the authors propose that an assemblage of aluminum minerals controls Al solubility. The solubility of iron is controlled by amorphous ferric hydroxide at all pH values.

In a stream receiving acidic effluents from a mine tailings deposit, suspended particulate matter formed as pH increased from 3.5 to 6.5 due to mixing with groundwater of pH greater than 7 (Karlsson et al., 1988). The particulate material consisted of  $\text{FeOOH} + \text{Fe}(\text{OH})_{3(\text{am})}$  and  $\text{AlOOH} + \text{Al}(\text{OH})_{3(\text{am})}$  and silicates and was often associated with organic matter. By the time pH was greater than 6.5 most of the iron and aluminum from the mine effluent and most of the total metal content in the aqueous phase precipitated in the particulate phase. About 50% of the Mn, Cu, Zn, Cd and Pb precipitated with the particulate phase, mainly by adsorbing on the hydrous oxide precipitate. Mn and Pb also coprecipitated to some extent. These processes gradually removed these metals from the aqueous phase with increasing pH. Under ice-covered anoxic conditions some of the Fe stayed in the divalent state reducing the amount of suspended matter that formed.



Eh-pH diagram showing contours of dissolved iron

After Drever, 1988

Figure 5

Acidophilic algae and bacteria from low pH environments have anionic cell walls and can concentrate aqueous dissolved metals onto cell walls and intracellular sites (Mann et al., 1990). Amorphous Fe and Ti concentrated at cell walls are bioprecipitated into microcrystalline aggregates of Ti- and Fe-oxyhydroxides which then act as scavengers for heavy metals such as Cu, Pb, Zn, Ni, Cd and Th. The formation of the oxide minerals by inorganic means is generally kinetically inhibited, so the biological catalysis of their nucleation plays a major role in determining the aqueous metal concentrations in acid mine drainage. Iron-oxidizing Thiobacilli can tolerate 0.37M aluminum, 0.15M zinc, 0.17M cobalt, 0.15M manganese, 0.16M copper, 0.1M chromium and 0.01M uranium. Lower tolerances are found for silver ( $10^{-9}$ M to  $10^{-5}$ M), mercury (0.05M) and molybdenum (0.03M). The bacteria can not tolerate oxides of selenium, tellurium and arsenic (Lundgren and Silver, 1980).

The general conclusion that can be drawn about the aqueous geochemistry of trace metals is that in most natural waters they are adsorbed out of solution at high pH due to adsorption on iron oxyhydroxides, but remain in solution at low pH. Microorganisms can act as scavengers for some heavy metals. Studies are needed in the existing water-filled pits before generalizations about the short and long term fate of trace metals in pit water can be made.

## Arsenic Speciation

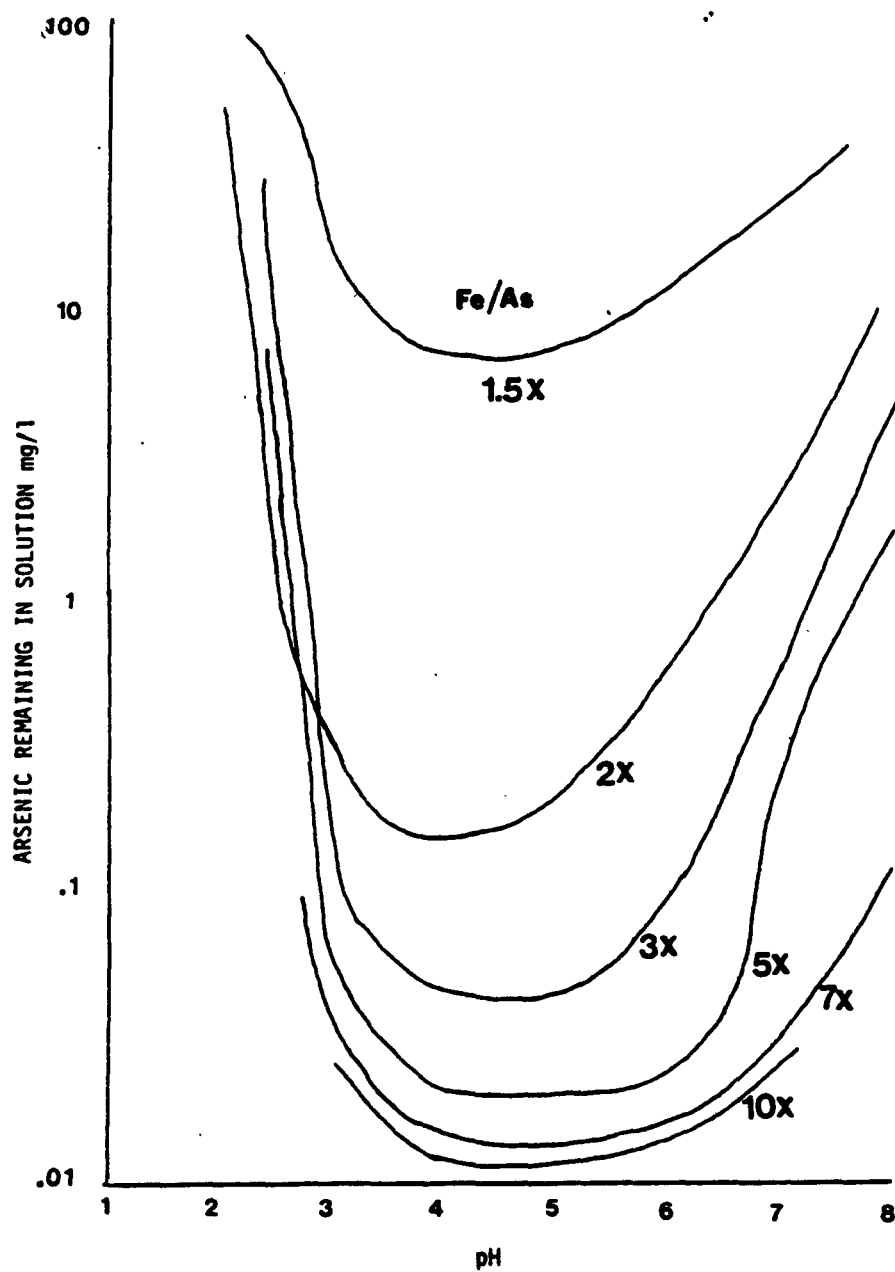
Naturally occurring arsenic is found in groundwater in many parts of the western United States, with moderate to high concentrations being common. Some of the highest concentrations are associated with mining (Welch et al., 1988). Arsenopyrite ( $\text{FeAsS}$ ) and arsenic-bearing pyrite are often found with gold ore. Other sources of arsenic include orpiment ( $\text{As}_2\text{S}_3$ ) and realgar ( $\text{AsS}$ ) and arsenic-rich iron oxides.

Arsenic has multiple valence states, and can form over 245 mineral compounds making it difficult to evaluate because its behavior is complex (Lynch, 1988). Weathering of rocks containing arsenic usually forms soluble arsenates with arsenic in the +5 oxidation state (Boyle and Jonasson, 1973). Under anoxic conditions arsenic is usually found in the +3 state. In general, soluble As(III) compounds are more toxic than As(V) compounds and inorganic As compounds are more toxic than organic As compounds (Bitton and Gerba, 1984).

The major processes that control arsenic concentration and speciation include mineral precipitation and dissolution, solution composition, adsorption and desorption, ion exchange, competing and complexing ions, chemical transformations, biologic activity, pH, Eh, aquifer mineralogy, and reaction kinetics (Welch et al., 1988).

Amorphous ferric hydroxide has an extremely high adsorptive capacity for arsenic (Pierce and Moore, 1982). However, the presence of materials such as humics, surfactants, polyelectrolytes, silicic acid, phosphoric acid, silica particles, feldspar and montmorillonite accelerate the growth of  $\text{FeOOH}$  crystals from amorphous ferric hydroxide and can result in the release of arsenic (Robins et al., 1988).

For effective removal of arsenic, the Fe/As mole ratio must be significantly greater than one (Robins et al., 1988). Figure 6 shows arsenic remaining in solution versus pH, and curves for increasing iron to arsenic ratios. More arsenic is removed from solution with higher iron concentration due to adsorption on iron hydroxide and coprecipitation forming minerals such as scorodite ( $\text{FeAsO}_4 \cdot \text{H}_2\text{O}$ ).



Arsenic removal from 300 mg/l As solution with six different levels of Fe/As.

After Robins et al., 1988  
Arsenic Metallurgy Fundamentals  
and Applications

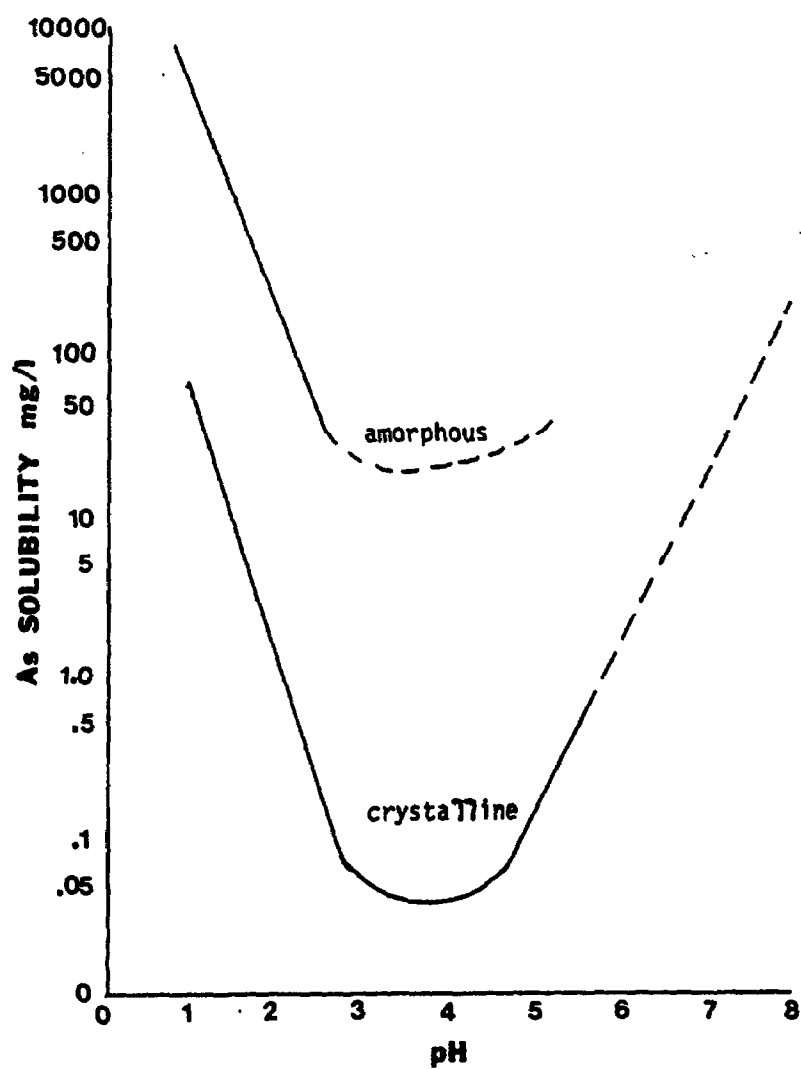
Figure 6

The solubility of scorodite is lowest at pH of about 4 (Figure 7) (Krause and Ettel, 1989). When scorodite forms, arsenic is removed from solution. On an Eh-pH diagram (Figure 8), the stability field for scorodite occurs in the high Eh, or oxidized range. Others have also shown that arsenic solubility and mobilization are minimized in oxidized conditions with pH of 4-5 (Robins et al., 1988; Masscheleyn et al., 1991).

Ferguson and Gavis (1972) described the arsenic cycle in a stratified lake. In the oxic epilimnion, reduced forms of arsenic are oxidized to arsenate which coprecipitates with ferric hydroxide. Some arsenate is transported across the thermocline by turbulent dispersion and convection. In the anoxic hypolimnion arsenate is reduced with the specific reduced arsenic species formed depending on the sulfur concentration and the Eh.

When a lake is mixing, arsenic adsorbs on iron oxides on the surface of the sediments and coprecipitates with hydrous iron oxides (Aggett and O'Brien, 1985; Mok and Wai, 1989). When the sediment becomes reducing due to burial or increased biological activity near the sediment/water interface, scorodite becomes soluble and Fe(III) is reduced to Fe(II) which is soluble at neutral pH's, so arsenic is released. The reduced iron and arsenic then can diffuse upward until they are introduced to the lake water, or they can be reprecipitated due to oxidation. During thermal stratification with anoxic conditions in the epilimnion, mobilization of arsenic from the sediments occurs (Crecelius, 1975; Mudroch and Clair, 1986).

Thermodynamic calculations predict that at equilibrium in oxic water As(V) should be the only stable oxidation state while As(III) should be the stable form in anoxic systems. However, arsenate and arsenite concentrations versus depth in a permanently stratified lake did not reflect the expected thermodynamic equilibria (Seyler and Martin, 1989). While As(V) was the dominant species in oxic waters, As(III) was also present in significant amounts. In anoxic waters, As(III) was dominant but As(V) was still present. This suggests

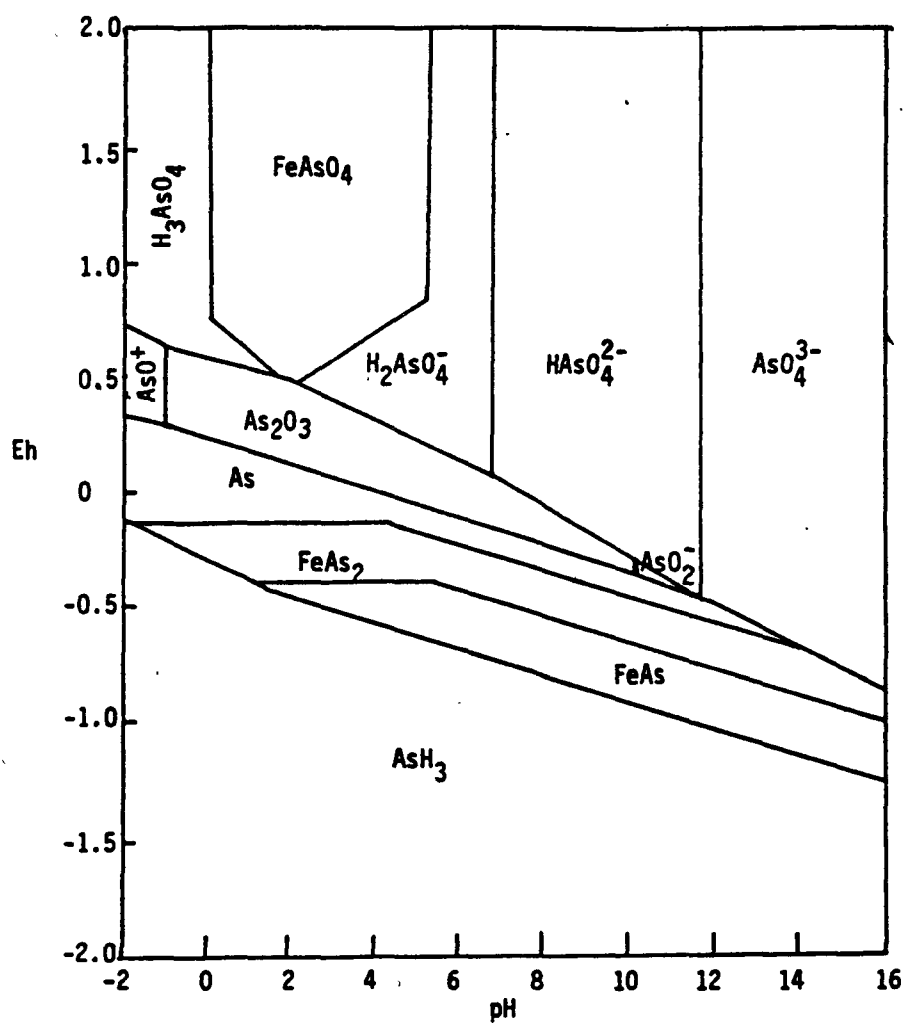


SOLUBILITY OF SCORODITE,  $\text{FeAsO}_4 \cdot 2\text{H}_2\text{O}$ , at 25°C.

after Krause and Ettel, 1989  
Hydrometallurgy, v. 22

Figure 7





Eh-pH diagram for the system Fe-As-H<sub>2</sub>O  
showing arsenic compounds at 25°C.<sup>2</sup>

After Mirza et al., 1988

Figure 8

that the reduction rate from As(V) to As(III) may be slow, causing incomplete response of the As redox couple to reducing conditions. Similar disequilibrium speciation has been shown for arsenic in the Baltic Sea (Andreae and Froelich, 1984). One possible explanation of the disequilibrium is that As(V) is reduced to As(III) by acting as an effective electron acceptor in microbial mineralization of organic matter and that Fe(III) competes for the role of electron acceptor, thereby decreasing the rate of As(V) reduction (Masscheleyn et al., 1991).

Disequilibrium speciation of arsenic can also occur due to changes in pH. A diurnal pH cycle due to photosynthesis in a perennial stream contaminated with arsenic caused a similar diurnal cycle in arsenate concentration, although the arsenate cycle lagged behind the pH cycle by several hours (Fuller and Davis, 1989). This indicates that the sorption-desorption kinetics of arsenic are also complex and slow, resulting in disequilibrium.

In addition to affecting disequilibrium, biologic organisms also play other roles in the speciation and solubilization of arsenic. Luong et al. (1981) showed that the presence of Thiobacillus ferrooxidans increases the rate at which arsenic leaches from gold-bearing material at least four fold over the abiotic leaching rate. Woolfolk and Whiteley (1962) and Johnson (1972) showed that bacteria can reduce arsenate to arsenite. Wakao et al. (1988) identified arsenic oxidizing bacteria that were aerobic and grew best between pH 3 and 4. McBride and Wolfe (1971) identified a strain of bacteria that could reduce and methylate arsenate to dimethylarsine under anaerobic conditions. Methylation of arsenic significantly reduces its toxicity (Eisler, 1988a). Anderson and Bruland (1991) showed that methylated arsenic is an important constituent in lakes. In a seasonally anoxic lake, dimethylarsenic acid became the dominant form of dissolved As within the surface photic zone during late summer and fall. When the lake turned over in early December methylated arsenic decreased while arsenate increased, suggesting the dimethylarsenic acid degraded to arsenate. Anderson and Bruland (1991) studied many lakes in California and Nevada and found methylated arsenic in all lakes except one that was highly alkaline (Mono Lake).

The speciation and the solubility of arsenic in pit water depends on the chemistry of the solution, redox state, pH, sorption and desorption, disequilibrium redox kinetics, and biological contributions, among other factors. At or close to neutral pH's, arsenic will

generally be soluble under anoxic conditions and will generally adsorb or coprecipitate with iron under oxic conditions. The disequilibrium in redox kinetics causes arsenic to be present in both the oxidized and the reduced form under any dissolved oxygen conditions. Optimum removal of As species from solution occurs at pH values of about 4 to 5. Excess iron relative to arsenic also minimized arsenic solubility. In general, oxygenated, non-alkaline conditions cause minimum solubility and mobilization of arsenic.

### **Evapoconcentration**

Because of the arid environment in which many mines in Nevada are situated, evapoconcentration may have a big effect on the water chemistry in the pit lakes. As an example of the magnitude of evaporation in Nevada, Big Soda Lake near Fallon in 1984 had an evaporation rate of 120 cm/yr and rainfall of only 9 cm/yr (Kharaka et al., 1984). Evaporation is a function of the vapor pressure above the water surface, the saturation vapor pressure corresponding to the temperature of the water surface, the wind speed above the water surface, wind shear and atmospheric stability. If the air in contact with the water is warmer and has more moisture than the ambient air, then the air at the water surface will be lighter and will tend to rise, creating an unstable condition. Evaporation will be greater with increasing instability. Wind shear is a function of the water temperature and fetch distance. Decreasing the water surface temperature increases wind shear and reduces evaporation. Increasing the fetch distance increases evaporation (Blee, 1988).

Geochemical modeling of evapoconcentration of pit water in Barrick Goldstrike's proposed Betze Pit resulted in an estimate of 35% increase in concentration of conservative elements by the year 2100. The lake will reach a hydrologic steady state in about 200 years with inflow from groundwater estimated at .05 m<sup>3</sup>/sec, outflow at .02 m<sup>3</sup>/sec and evaporation at .03 m<sup>3</sup>/sec. The lake will reach a chemical steady state sometime after it reaches a hydrologic steady state, and at that time concentrations of conservative elements will have increased by a factor of 2.25 (BLM, 1991).

In order to predict the effects of evapoconcentration in arid environment pit water lakes, information is needed about: the geometry of the drainage basins, local climate

averages, the water balances in the pits, and the geochemistry of the pit waters. A numerical simulation model that considers all of these factors and predicts the effects of evapoconcentration should be developed.

### **Hydrothermal Activity**

Nevada is located in a geologic area of high heat flow causing elevated groundwater temperatures. Much of the geothermal activity is associated with fault systems. Mineralization at mines is usually associated with fault systems too, so it follows that some mine sites will have thermal groundwater. Garside and Schilling (1979) outlined the locations of thermal waters throughout the state of Nevada.

The effects of hydrothermal activity on pit water lakes can be inferred from studies in the literature on hydrothermal inputs to natural lakes. Yellowstone Lake in Wyoming is oligotrophic and contains hydrothermal springs and gas fumaroles and has high geothermal gradients within the lake bed (Klump et al., 1988). The gases emitted by the fumaroles are primarily carbon dioxide with traces of methane and hydrogen sulfide. The hydrothermal waters (up to 70°C) are anoxic and high in dissolved nutrients and major ions. Mats of microbial heterotrophs and photo- and chemolithotrophs and dense congregations of aquatic worms, zooplankton and sponges were found surrounding some of the vents.

Instability of water in lakes is caused by differences in density, and temperature affects density. At atmospheric pressure, water is at maximum density at 4°C. However, the temperature of maximum density decreases with increasing pressure. Eklund (1963,1965) defined the temperature of maximum density,  $T_{MD}$ , for a specific volume of fresh water,  $V$ , as a function of pressure or depth as follows:

$$\begin{aligned}
V &= V_o[1-a+bO]P + cO^2 \\
T_{MD} &= 4^\circ\text{C} - O', \quad O' = (b/2c)P = (.021^\circ\text{C}/\text{bar})P \\
&= (2.1 \times 10^{-3}\text{C}/\text{m})z \\
P &= \text{water pressure at depth } z \\
a &= 49.458 \times 10^{-6}/\text{bar} \\
b &= .327 \times 10^{-6}/\text{bar}^\circ\text{C} \\
c &= 7.8 \times 10^{-6}/^\circ\text{C}^2 \\
O &= (4.00 - T)^\circ\text{C}
\end{aligned}$$

Osborn and LeBlond (1974) and Chen and Millero (1977) developed a test of static stability of water in lakes, taking into account the vertical variation of temperature,  $T$ , and salinity,  $S$ , as follows:

$$\begin{aligned}
2V_o c(T-T_{MD})(dT/dz - dT_A/dz) &< (dV/dS)P(dS/dz) \\
dT_A/dz &= \text{adiabatic temperature gradient} = \\
&T(2V_o c p g / c_p)(T-277.16 + p g z b / 2c) \\
T &= ^\circ\text{K at depth } z \\
p &= \text{mean water density above} \\
c_p &= \text{specific heat of water at constant temperature}
\end{aligned}$$

Golubev (1978) showed that water temperatures in Lake Baikal at depths greater than 250 m are always less than  $4^\circ\text{C}$ . The bottom water (300-400 m deep) is less than  $3.5^\circ\text{C}$ . Golubev found a localized region in Lake Baikal with a 1-2 m thick bottom water layer with a temperature at least  $.5^\circ\text{C}$  warmer than the overlying water. This represents a very unstable heat flux due to hot spring discharge.

Williams and Von Herzen (1983) found a similar situation in Crater Lake, Oregon. At about 295 m depth the temperature everywhere is less than  $4^\circ\text{C}$ . Below 295 m the deep water temperatures increase with increasing depth. Thermal springs discharge into the deep water and affect the temperature stratification, circulation and chemistry of the lake. Williams and Von Herzen (1983) described thermal fluids discharging from a spring as fluid rising in a plume and rapidly mixing the deep lake waters. The rest of the water is dense due to dissolved solids and flows down slope where it mixes turbulently with the overlying lake water. Some of the water ponds on the lake floor where the mixing process is considerably slower. The authors determined that the most important effect of the high heat input at Crater Lake is that it caused the lake to mix vertically on a time scale of weeks to months.

Williams and Von Herzen (1983) said that Crater Lake might be meromictic due to an increase in suspended solids with depth, although the chemistry does not change with depth. Thermal springs discharge about  $6.35 \times 10^9$  g of dissolved solids into the lake annually. The authors describe the monimolimnion as well-mixed due to thermal convection. They said that the deeper monimolimnion is not strongly influenced by periods of circulation and mixing in the overlying mixolimnion, although it is not stagnant. The uniform chemical data show that Crater Lake's waters are well-mixed laterally and vertically, so there must be significant mixing between the mixolimnion and the monimolimnion. Williams and Von Herzen (1983) describe the maximum density point as a natural top for the thermal convection system of the monimolimnion. As thermal fluids rise (because they are less dense than the surrounding fluids), they become diluted and cooled by the colder lake water. As they cool they approach the temperature of maximum density at which they become negatively buoyant. Williams and Von Herzen (1983) concluded that the standard limnologic definitions do not consider the effects of terrestrial heat flow on the heating of deep lake water.

Hydrothermal input to pit water lakes will affect the temperature, stratification, circulation and chemistry of the lakes. Mixing may be enhanced due to thermal convection or suppressed by the input of dissolved solids which increase density. The elevated temperatures will affect chemical speciation, if they are high enough, but only close to the thermal springs because the spring water will cool as it is assimilated into the surrounding water. Additional studies are needed on other existing lakes containing thermal springs in order to better quantify the range of potential effects of the springs and especially to determine how they affect circulation of water in the lakes.

## **Existing Pit Lakes**

Examining some case studies of existing pit lakes can provide additional information about the factors that affect pit water quality. Because precious metal open pit mining below the water table is a relatively new phenomenon, most mines are still in production and dewatering. Only a few such pit lakes exist. However, there are examples of water-filled open pits from mining of other minerals such as uranium, phosphate, coal and copper. Although different factors may affect pit lakes resulting from mining of different mineral commodities, information relevant to understanding conditions in precious metal pit lakes can be gleaned from other types of pit lakes.

### **Phosphate Mines**

Many phosphate mine pit lakes are found in the state of Florida. As mining methods changed, the shape of the lakes changed. Phosphate pits dug before 1920 were relatively circular. This shape did not allow much littoral development, yet the lakes were productive and supported large populations of game fish. The 1920's introduction of the dragline allowed parallel excavations, 60-90 m wide, 600 m long and up to 20 m deep. Rules for the reclamation of water bodies created by mining in Florida were intended to encourage the development of lakes having shapes similar to natural lakes. These characteristics include shallow basins, a well-developed littoral area, and a large portion of the water column in the euphotic zone. Specifically, the regulations require (Florida Dept. of Natural Resources, 1975, cited by Pratt et al., 1985) that bottom slopes within 8 m of shore must not be steeper than 4:1 (horizontal to vertical). To encourage littoral vegetation, at least 25% of the lake surface must be within the zone of water fluctuation, or adjoining wetlands must be created. To provide for fish bedding areas and submerged vegetation zones, at least 20% of the surface must fall within a zone between the annual low water line and the minus 2 m annual low water (Pratt et al., 1985). Florida's reclamation regulations also require trees along 50% of the perimeter of reclaimed lakes (Ericson and Mills, 1986). Trees could provide nutrients to aquatic biota in the lakes by dropping organic matter into the water. To allow for the possibility that precious metal mine pit lakes could be used as fisheries, plans could be made to leave benches at the proper elevation to provide littoral areas in the future pit lakes.

Pratt et al. (1985) studied reclaimed phosphate pit lakes ranging in age from less than one year to seven years and unreclaimed lakes over sixty years old. They found reclaimed lakes to be dynamic systems resembling newly-formed reservoirs. These lakes tend initially to be eutrophic with high concentrations of phosphate, nitrogen and trace minerals. Phosphate pit lakes do not contain food webs as complex as those found in natural lakes because biological introductions to the new lakes tend to be haphazard. Also, the biogeochemical conditions in the lake are constantly changing during the first several years prior to stabilization and this may pose a challenge to organisms. Precious metal mine pit lakes will also receive haphazard biological introductions. Unlike in phosphate pits, phosphorous is likely to be a limiting nutrient in precious metal pit lakes, so biological productivity would be lower.

### **Uranium Mines**

With uranium mines, there is often concern about radiation, so tailings are usually covered and the pits backfilled. Water quality is generally not suitable for any use. The Jackpile-Paguate uranium mine in west central New Mexico consists of three open pits which are steep sided and partially filled with water since mining stopped in 1982 (Reith et al., 1990). Water in each pit is slightly contaminated with trace metals, radionuclides, and suspended sediments. Levels of contamination did not necessitate undertaking a treatment program. Below-grade ore stockpiles were used as backfill in the pits. Radon barrier covers consisting of 30 cm of shale and 60 cm of soil were placed on the ore-derived backfill material.

The Nabarlek Uranium Project in the Northern Territory, Australia mined a small high-grade ore body (McLoughlin et al., 1988). The open pit was used for tailings disposal and contaminated water storage during milling operations. The pit had a net gain in water storage each year due to seasonal high rainfall, and seepage from the pit occurred. To prevent this, measures that exclude water were planned. After the tailings in the pit settle, the mound over the surface of the pit will be leveled and a layered capping of clay and rock will be placed, followed by topsoiling and revegetation.



Uranium mining has produced many open pits in South Texas. Many of these are now water-filled and about 30 m deep. Kallus (1977) found pit water to have alpha and beta sources of radiation and high concentrations of arsenic (average=.018 mg/l, maximum value=.041 mg/l), and selenium (average=.022 mg/l, maximum value=.054 mg/l). Kallus (1977) cited an unpublished master's thesis (Itin, 1975) that found these lakes to be unsuitable water sources for human consumption, recreation, irrigation, stock and wildlife watering and for fish and aquatic life. Itin (1975) found that the pit lakes were not thermally stratified.

The water in the Whites Pit at the Rum Jungle Uranium Mine in Australia had a pH of 4.75 in 1959, one year after mining ended, and a sulfate concentration of 180 mg/l. In 1974, after the addition of unneutralized tailings to the pit, the pH had decreased to 2.4 with sulfate concentrations at 9000 mg/l. In 1974, the Intermediate Pit had a pH of 3.5 and a sulfate concentration of 2000 mg/l. The Whites Pit lake is stratified with dissolved oxygen concentrations of less than 1 mg/l below 5 m (Goodman et al., 1981). A microbial study of the Rum Jungle pits showed that the largest populations of Thiobacillus ferrooxidans occurred in the sediments of the Whites Pit, indicating that they may be oxidizing sulfides anaerobically, using something besides oxygen (such as iron) as a terminal electron acceptor. Remediation at Rum Jungle is discussed in the Reclamation section of this paper.

### Coal Mines

Both thermally stratified and non-stratified conditions exist in surface coal mine lakes (Voelker, 1985). A coal pit water impoundment in Montana did not thermally stratify until the fourth year after filling (Goering and Dollhopf, 1981). Lack of stratification in the first three years may have resulted from larger inflow which produced more turnover and mixing. Heat absorption by turbidity caused by reddish-black ferric oxides played a major role in summer thermal stratification of coal strip mine lakes in Missouri (Parsons, 1977). Heat budgets for turbid lakes were 65% less per unit volume than for clear lakes which did not stratify.

Acidified coal strip mine lakes go through a series of successional stages in which the acid, sulfate, Ca, Mg, Al, and Fe concentrations are gradually reduced (Campbell and Lind,

1969). This recovery has been referred to as the slow "titration" of the acidity by sulfate reducing bacteria (Doyle, 1976). Decker (1971) showed that adding primary sewage sludge added sufficient organic matter to greatly accelerate the recovery process. Sulfate reducing bacteria utilize organic matter and release sulfide ions, which either combine with metal cations and precipitate or combine with  $2\text{H}^+$  at low pH to form  $\text{H}_2\text{S}$  gas which is released to the atmosphere (Doyle, 1976).

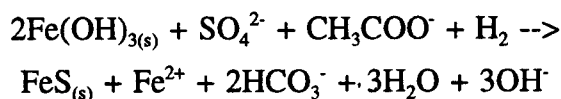
Acid mine water increases the rate of weathering of clay minerals, feldspars and carbonates. Aluminum silicates produced by this mechanism tend to buffer pH (Kelly, 1988). King et al. (1974) studied the restoration of acid lakes and found that below pH of 4.5 neutralization proceeded slowly. Once the aluminum buffer system was consumed then neutralization was more rapid. In the pH range of 4.5 to 5.0 bacteria are no longer limited by harsh acidic conditions so sulfate reduction occurs at a constant maximum rate (Doyle, 1976).

Concentrations of  $\text{CO}_2$  and  $\text{H}_2\text{S}$  gases also increase as the bacteria growth rate increases. Above pH = 6.4 (the  $\text{pK}_1$  for carbonic acid) bicarbonate and hydrogen ions form by dissociation:



This is a buffering system which maintains the pH near 6.4. Other buffers include  $\text{HSO}_4^-$ ,  $\text{H}_2\text{CO}_3$ ,  $\text{H}_2\text{S}$  and ionic metals (especially  $\text{Al}^{3+}$ ). Each lake is unique since buffers will be present in different concentrations and the initial amount of acid present will also vary. When the pH is high enough for dissociation of carbonic acid from  $\text{CO}_2$ , the bicarbonate alkalinity system is established and phytoplankton and zooplankton can then form the base of the food chain for the aquatic community. After an acidified lake has recovered, its water chemistry is like that of an early eutrophic lake except that the sulfate:bicarbonate ratio is high (Doyle, 1976).

Authigenic inorganic reduced sulfur minerals (mostly  $\text{FeS}_2$  and elemental S) are found in the sediments of final-cut coal strip mine lakes (Wicks et al., 1991). The reaction for sulfate reduction and precipitation of sulfides follows:



The products of this reaction,  $\text{HCO}_3^-$  and  $\text{OH}^-$ , are bases. In the strip mine lakes pore water iron concentrations did not correlate with sediment sulfur content. Canadian Shield lakes, however, had a strong pore water iron/sediment sulfur correlation (Carignan and Tessier, 1988). Wicks et al. (1991) attributed this difference to the availability of iron. The Canadian Shield lakes are located on crystalline metamorphic bedrock overlain by glacial deposits, so the source of iron is probably the weathering of crystalline bedrock. This means that a resistant mineral is the source of the iron. Iron is released slowly, so as soon as dissolved iron is delivered to the sediments it is consumed by the reaction. Conditions in the Canadian Shield lakes are iron-limiting. The strip mine lakes, on the other hand, were located on sandstone, shale and limestone bedrock. The source of iron is dissolved Fe in the pore water of the sandstone and shale and very reactive  $\text{Fe}(\text{OH})_3$  in the sediments of the overburden. The sulfur mineral formation process is not iron or sulfate limited in strip mine lakes where reactive iron and sulfate are abundant. Instead, the type of, and amount of, organic matter controls the formation of reduced sulfur minerals.

The following can be learned from these coal lake studies that may pertain to precious metal pit lakes:

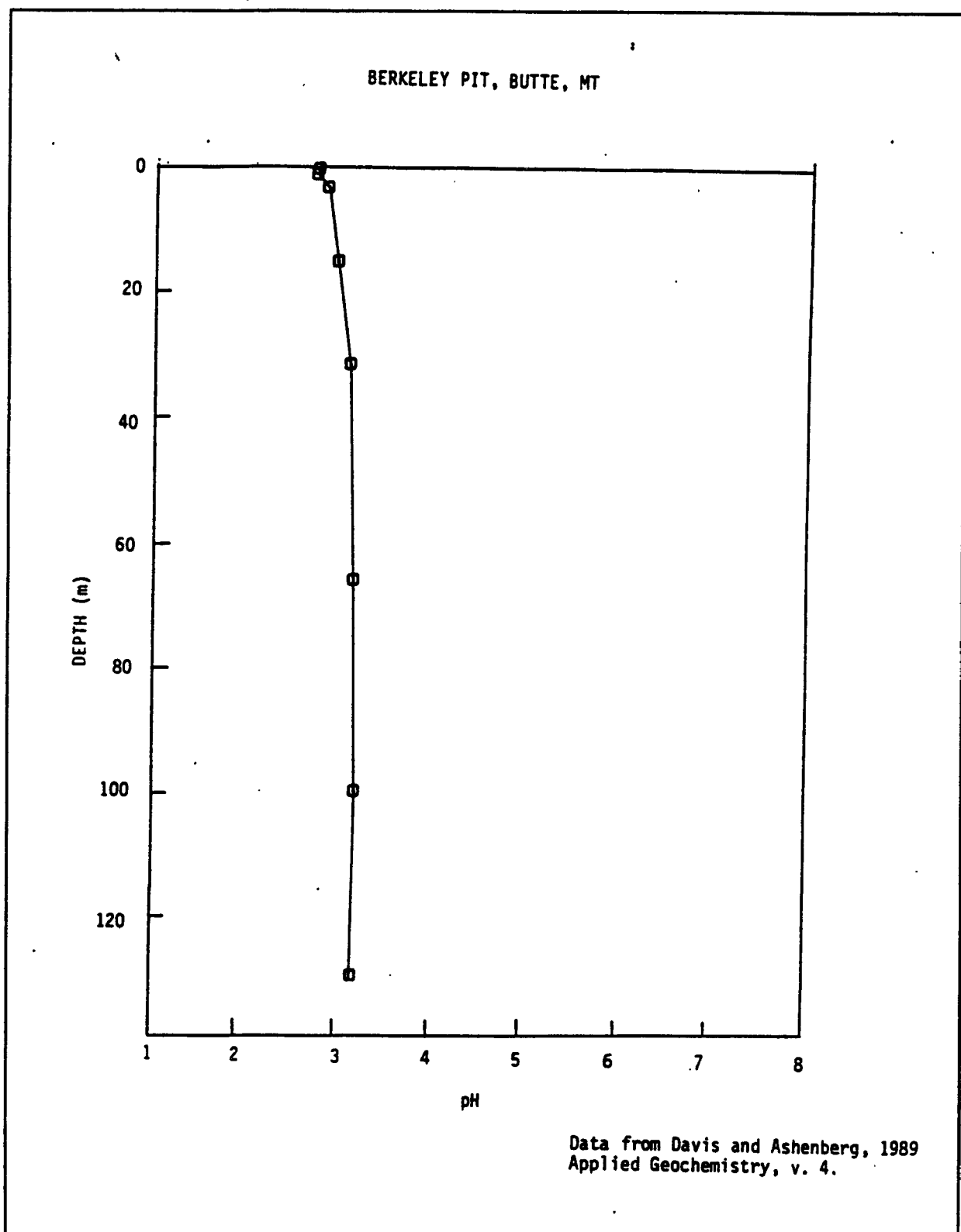
- stratification is unlikely during major inflow of water to the pit,
- turbidity caused by the precipitation of authigenic ferric oxides can change the heat budget in a pit lake and cause stratification,
- bacterial sulfate reduction can aid the recovery of acidified lakes by removing potentially toxic metals and hydrogen ions, and
- the rate of sulfate reduction can be limited by amount of organic matter in lakes in sedimentary terrains, and limited by iron concentration in crystalline rock terrains.

### **Copper Mines**

Mining of the Berkeley Pit porphyry copper mine in Butte, Montana ended in 1982 and flooding of the pit began in 1983. The pit dimensions are 1.8 km by 1.4 km across and 542 m deep. In 1987 the water level in the pit was rising at a rate of 22 m per year. Davis and Ashenberg (1989) estimate that the pit will overflow during the year 2009 and will

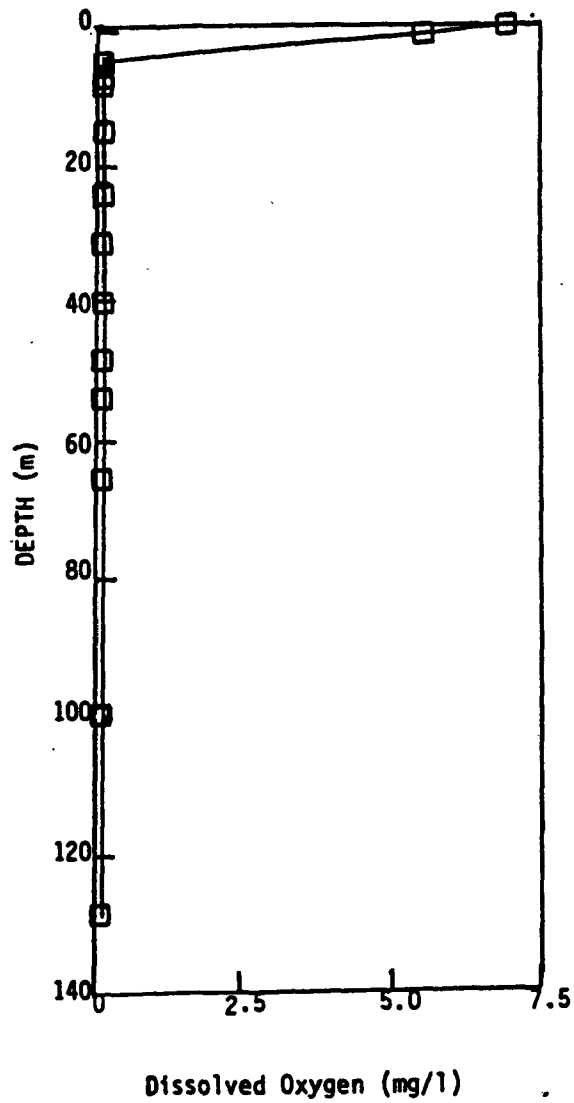
intersect the alluvial aquifer by 1996. In October of 1987, geochemical sampling was completed in a depth profile down to 130 m in the middle of the pit lake in an attempt to characterize the aqueous solution in the pit and to define processes which may control metal concentrations and distribution (Davis and Ashenberg, 1989).

The pH of the Berkeley Pit water ranges from 2.7 at the surface to 3.17 at depth (Figure 9). Davis and Ashenberg (1989) attribute the lower pH at the surface to surface water runoff. Huang and Tahija (1990) showed that the surface water source is an active tailings pond and leach pads and that these surface sources supply most of the trace metal ions to the pit. Dissolved oxygen decreases exponentially from 0 to 3 m, and below 3 m the system is suboxic (Figure 10). Within the top 5 m most of the ferric iron is reduced and Fe(II) becomes dominant (Figure 11). Ferric iron was below detection limits from 25 to 100 m depth. Arsenic concentrations were low in the upper 15 m but increased with depth below 15 m (Figure 12). At the same depths that all of the iron is reduced, most of the arsenic occurs in the +5 oxidation state. This is because at pH of 3, iron is reduced at an Eh of about .75 V, while arsenic is reduced at an Eh of about .4 V. The field Eh profile for the Berkeley Pit (Figure 13) goes from about .8 V at the surface to about .45 V at 130 m. The mass of solids on the filter (Figure 14) is a measure of total suspended solids which increase



**Figure 9**

BERKELEY PIT, BUTTE, MT



after Davis and Ashenberg  
1989, Applied Geochem., v. 4.

Figure 10

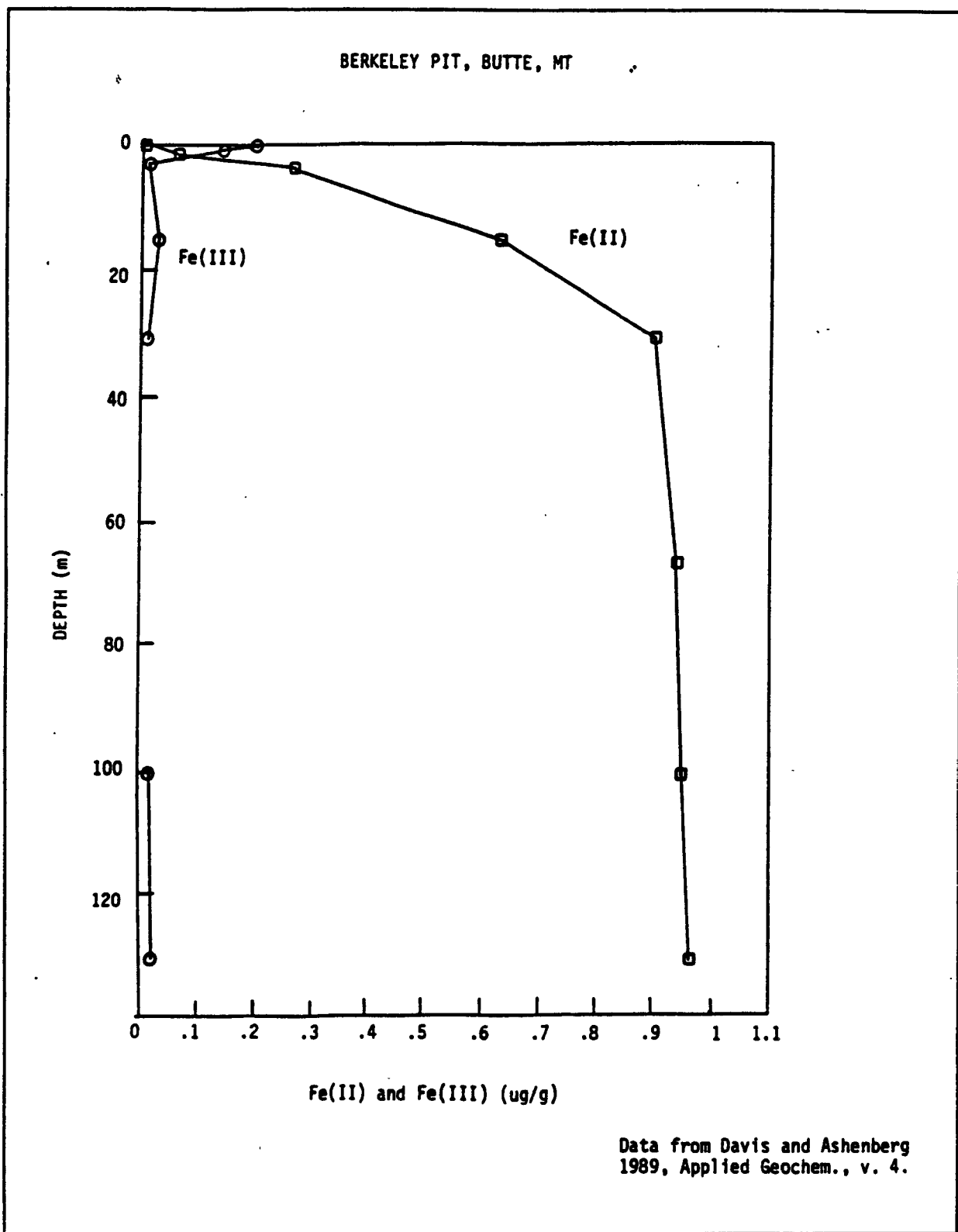


Figure 11

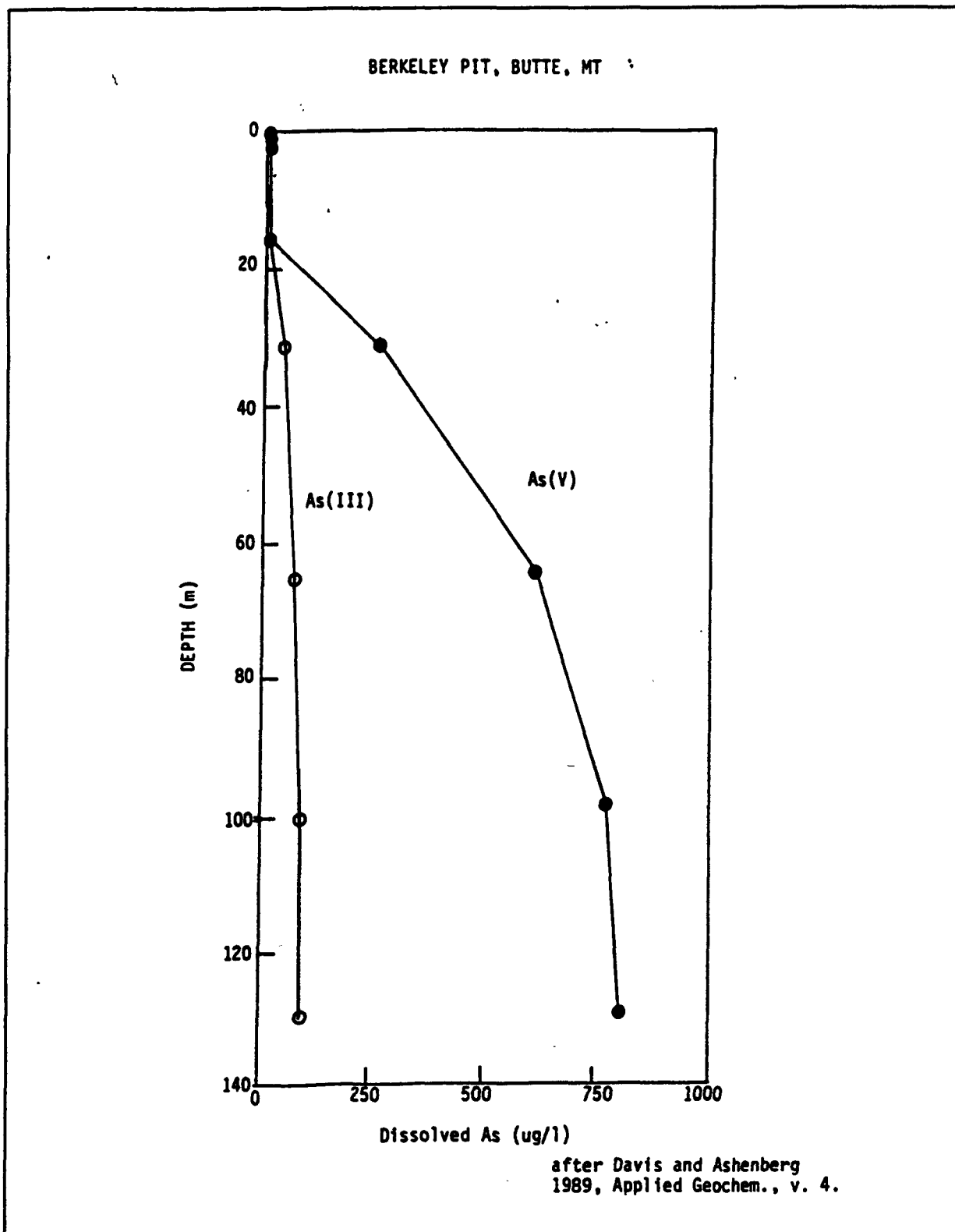
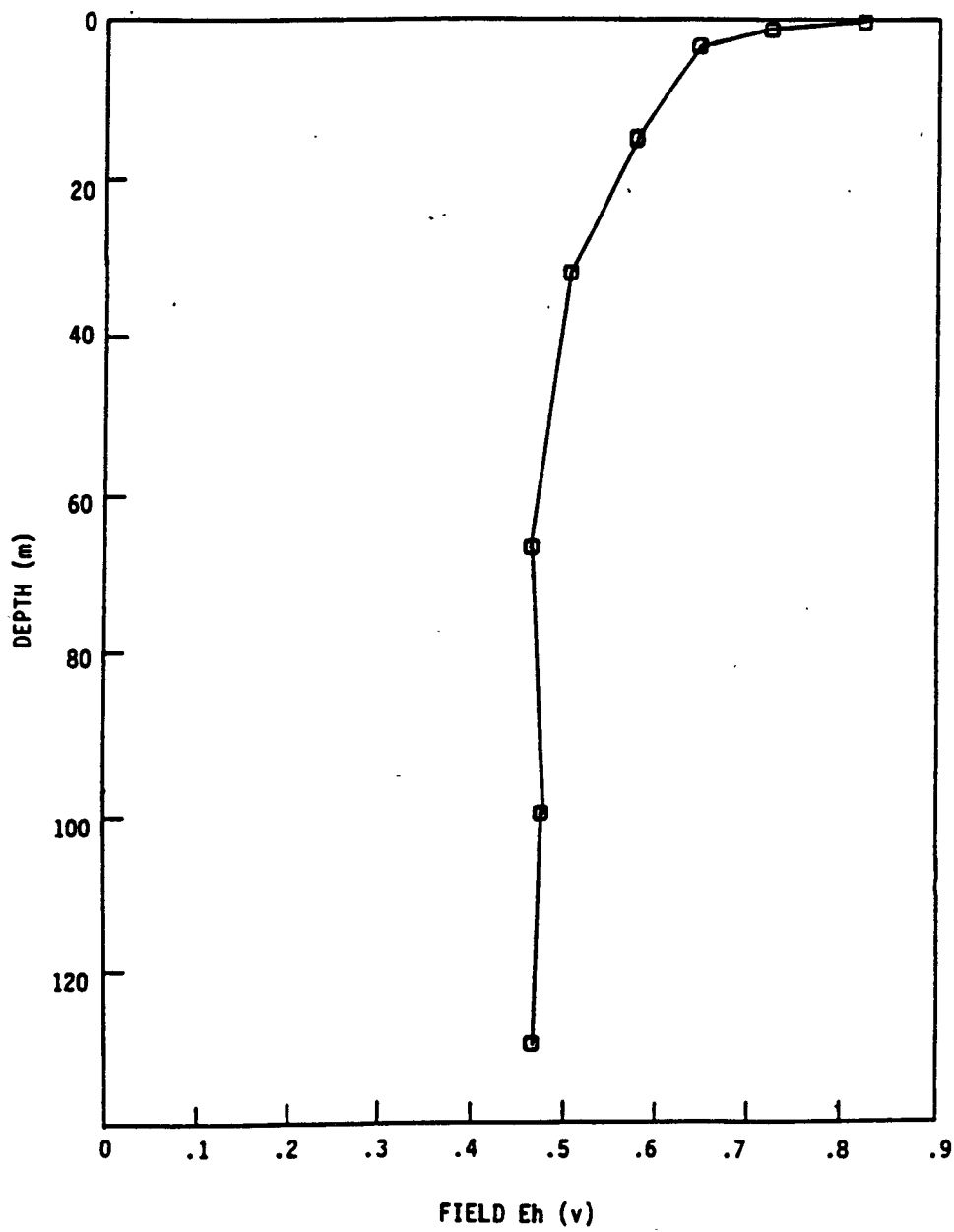


Figure 12

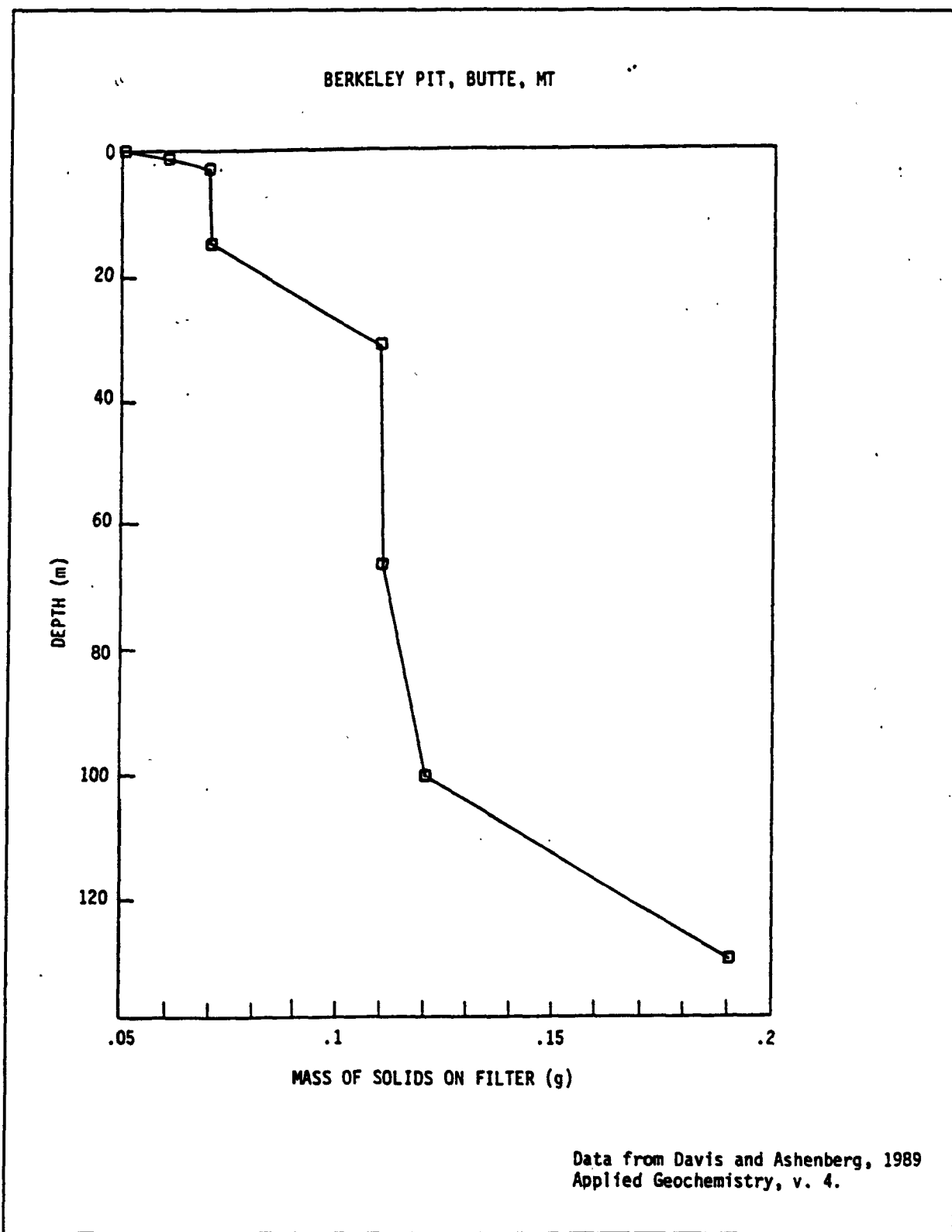


BERKELEY PIT, BUTTE, MT



Data from Davis and Ashenberg, 1989  
Applied Geochemistry, v.4.

Figure 13



**Figure 14**

TABLE 2

## Pit Water Quality at Ruth

	Liberty 8/2/91 (mg/l)	Ruth 9/30/86 (mg/l)	Ruth 8/2/91 (mg/l)	Kimbley 9/24/91 (mg/l)
Arsenic	<0.001	<0.004	<0.001	<0.180
Barium	<.01	<0.3	0.02	0.009
Cadmium	0.412	0.148	<0.005	<0.007
Chromium	0.03	0.1	0.02	<0.010
Fluoride	13	2.4	2.3	2.61
Lead	0.005	<0.01	0.001	<0.050
Mercury	<0.0002	<0.0001	0.0002	0.838
Nitrate	<0.1	0.02	2.4	<1.0
Selenium	<0.002	<0.004	0.04	<0.130
Silver	<.01	<0.01	0.04	<0.020
Chloride	30	6.6	44	264
Copper	50.6	31	17.5	0.172
Iron	20.7	2.94	0.17	0.455
Manganese	77	108	0.611	0.31
pH	2.86	3.23	8	7.59
Sulfate	2860		1330	1607
TDS	3480	2130	1820	3310
Zinc	35	40	0.041	2.43

Woodward-Clyde Consultants, 1992

**TABLE 3**  
**Yerington Pit Water Quality**

	<u>7/26/90 (mg/l)</u>	<u>10/29/90 (mg/l)</u>
pH	8.06	8.21
TDS	638	628
Alkalinity: Total	143	134
Alkalinity: HCO <sub>3</sub>	117	110
Ca	49	230
Mg	14.3	22.3
K	16	6.9
Na	48.7	74
Cl	43	40
F	1.7	1.4
NO <sub>3</sub> as N	0.67	<0.5
SO <sub>4</sub>	240	242
As	<.002	0.014
Ba	0.042	0.034
Cd	<.002	0.008
Cr	0.004	0.02
Cu	0.731	0.232
Fe		0.581
Pb	0.011	0.012
Mn	0.09	0.076
Hg	<.002	<.001
Se	0.004	<.002
Ag	<.010	<.010
Zn	<.030	0.081

Source: NDEP, 1991

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TABLE 4

U.S. EPA Drinking Water Standards

<u>Constituent</u>	<u>Max. Contaminant Level (mg/l)</u>
Arsenic	0.05
Barium	1.0
Cadmium	0.010
Chromium	0.05
Lead	0.05
Mercury	0.002
Nitrate (as N)	10.0
Selenium	0.01
Silver	0.05
Fluoride	1.4-2.5 (based on ave. ann. tmp.)
TDS	500
Chloride	250
Copper	1.0
Iron	0.3
Manganese	0.05
Sulfate	250
Zinc	5.0
pH	6.5-8.5

---

The Brenda deposit in the Okanagan District in British Columbia is a molybdenum-copper porphyry. The pit is 300 m across and more than 90 m deep. The pH of this pit is 7.3 (McCandless, 1992). Adjacent tailings pond water contained 1.5 mg Mo/l, a concentration too high for use in irrigation, so the water was pumped into the pit. Dissolved Mo concentrations in the pit range from 1.37 mg/l at 1 m depth to 1.69 mg/l at 45 m.

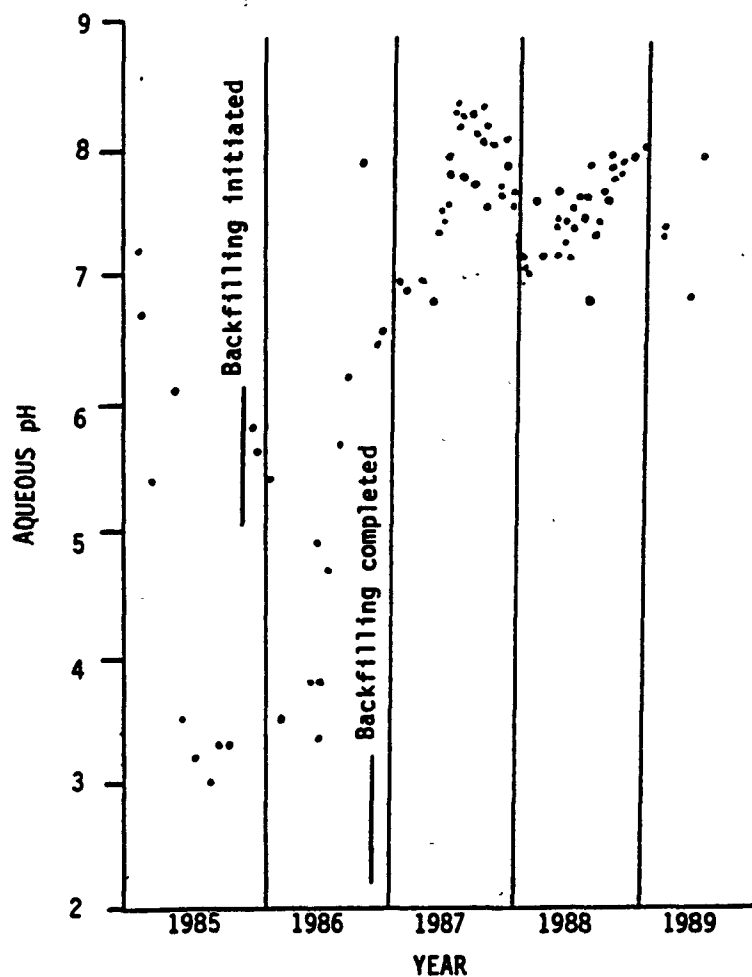
The War Eagle deposit in the Yukon Territory near Whitehorse is copper in skarn. The pit is 26 m deep and 100 m across, and it filled in 12 years, beginning in 1971. No acid was produced and the pH of the water is 7.8-8 (McCandless, 1992). Dissolved oxygen goes to 0 below 12 m suggesting minimal turnover.

The copper pit lakes referenced in this section show the range in water quality that is possible. They point out the importance of local geology and wallrock geochemistry in determining ultimate pit water quality.

#### **Precious Metals: Silver Mines**

There are three pits at Equity Silver Mines Ltd. in British Columbia. The Southern Tail Pit was mined first, and then was backfilled with material from the Main Zone Pit. Patterson (1990) described the backfilling process, which occurred in three stages. Backfilling to a horizon one meter below the projected flood plane was followed by placement of a two meter layer of inert non-acid producing waste to serve as a buffer zone. The third backfilling stage involved placing waste on top of the buffer layer and above the water table. Since these wastes were also acid generating they were reclaimed to reduce oxidation rates.

The pH of the Southern Tail Pit water from 1985 through 1989 is shown in Figure 15. The pit water was initially neutral and then pH decreased to 3 by mid-1985. Back-filling with material from the Main Zone Pit began in October, 1985. The Main Zone waste rock had reactive neutralizing minerals, and the Southern Tail pit water pH rose to almost 6 within



SOUTHERN TAIL PIT, EQUITY SILVER MINES, B.C.

After Morin, 1990  
Acid Drainage from Mine Walls  
The Main Zone Pit at Equity  
Silver Mines

Figure 15

a few months, then dropped again to below 3.4, increased to 7 by the end of back-filling in 1987 and varied from 7 to greater than 8 from 1987 to 1990. The development of acidic conditions as the pit began to fill may be attributed to the combination of flushing stored acid products on the walls and within the unsaturated fracture networks downward to the bottom of the pit and the decreasing flow of alkaline groundwater into the pit (Morin, 1990). Copper, iron and zinc concentrations versus pH in the Southern Tail Pit are shown in Figure 16. As pH increases, iron concentrations decrease due to precipitation of FeOOH and copper and zinc concentrations decrease due to adsorption on FeOOH, although there is a lot of variation in the zinc data.

In 1990 the Main Zone Pit was to be mined out at a depth of approximately 200 m at which time wastes from the third pit (the Waterline) would be backfilled. The pit would then be flooded to cover the wastes. A dam was to be constructed at the pit entrance to raise the water level so it would cover a portion of the wallrock that was acid generating.

#### **Precious Metals: Gold Mines**

The Nickel Plate Pit near Hedley, British Columbia contained a gold skarn deposit. The pit is 28 m deep, and less than 100 m across, and it filled in 3 years. No acid was produced and the pH of the water is 7.8-8 (McCandless, 1992).

The Cortez Gold Mine in Nevada is in the Roberts Mountain Formation which is limestone. This pit had an oxide ore body, and started filling with water in the early 70's. Water quality data is shown in Table 5. The pit is currently 20-30 m deep and in the early 80's bass were planted and still survive today. The area adjacent to this pit is being mined again so fishing is not permitted now, but it was allowed in the past. The fish are not fed by mine personnel, indicating that there is enough primary productivity in the pit to support a full food chain that sustains the fish. There are also reeds growing at the edge of this pit (List, 1992).



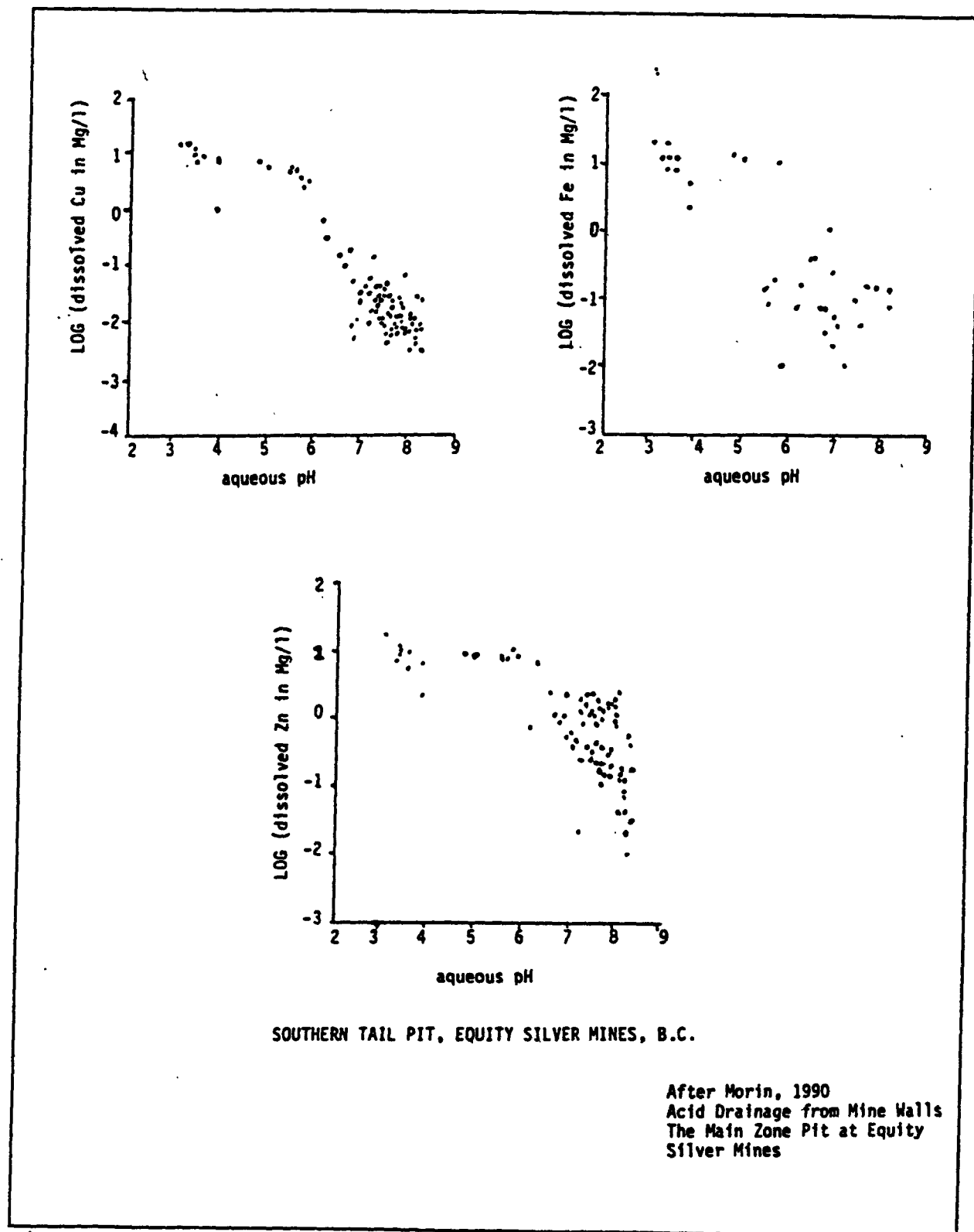


Figure 16

TABLE 5

Cortez Gold Mine Pit Water Quality  
sampled 9/13/90, unfiltered sample

Element	Conc. (mg/l)	Element	Conc. (mg/l)
Ag	0.005	Mo	0.013
Al	<<<	Na	68.100
As	<<<	Ni	<<<
Au	<<<	Pb	<<<
B	0.327	Pd	<<<
Ba	0.061	Pt	<<<
Be	<<<	S	30.500
Ca	45.100	Sb	<<<
Cd	<<<	Se	<<<
Co	0.004	Si	14.200
Cr	<<<	Sn	<<<
Cu	<<<	Sr	0.778
Fe	<<<	Te	<<<
Hg	0.081	Ti	0.012
K	11.500	Tl	0.053
Li	0.212	V	<<<
Mg	19.400	W	<<<
Mn	0.001	Zn	<<<

<<< indicates less than detection limit

M. List, Cortez Mine  
Personal communication, 1992

### Effects of Pit Water Quality on Life

The bass living in the pit lake at Cortez Gold Mine, without being fed by humans, indicate that there are enough naturally occurring nutrients in the pit water for them to survive. In addition, explosives used for excavating mines can later contribute nitrogen as nutrients (Morin, 1988). If all of the necessary nutrients are present in the pit water, the other concern for survival of aquatic biota is water quality and the concentration of elements that are toxic. It is difficult to determine safe levels of metals for aquatic organisms because the level at which a metal becomes toxic to a particular species depends on time of exposure, temperature, dissolved oxygen concentration, pH, salinity, hardness, water velocity, and the interaction between combinations of metals (U.C. Berkeley Mining Waste Study Team, 1988).

The effects of various aqueous pH levels on biota have been studied (Appalachian Regional Commission, 1969; Bell, 1971). Also, the effects of elevated concentrations of various elements on fish, wildlife and invertebrates have been reviewed (Eisler, 1985a, 1985b, 1986, 1987, 1988a, 1988b, 1989, 1990).

Bell (1971) tested mature larvae and nymphs of 9 species of aquatic insects (dragonflies, stoneflies, caddisflies and mayflies) in the laboratory for their tolerance to low pH. All of the tested species have high value as fish food. In general, Bell found that caddisflies are very tolerant of low pH (30-day TL 50 as low as pH = 2.45). The 30-day TL 50 is the pH at which 50% of the organisms died after 30 days. Stoneflies and dragonflies are moderately tolerant (pH 3.71 to 5.0), and the mayflies are sensitive (30-day TL 50 at pH = 5.38). The more sensitive insects will be limited in numbers and species composition under prolonged acid conditions. In addition, Bell found that under low pH conditions the percentage of aquatic insects which emerge successfully also decreases. The pH at which 50% successful emergence takes place ranges from 0.52 to 2.10 pH units higher than the 30-day TL 50 value for the species tested. These aquatic insects were generally more tolerant than fish however. The Appalachian Regional Commission (1969) said that most data indicate that fully developed adult fish can live in waters of pH ranging from 5.0 to 9.0. Below pH of 5.0 the productivity of aquatic ecosystems is considerably reduced.

Copper, zinc and cadmium accumulate in fish livers in a portion of the Sacramento River that receives acid drainage from mining areas (Wilson et al., 1981). The metals did not

accumulate in fish flesh. Aqueous metal concentrations at one sample location were .051 mg Cu/l, .214 mg Zn/l, and .0023 mg Cd/l. LC-50's for various aquatic organisms range from .0006 to 250 mg/l for cadmium, from .055 to 60.2 mg/l for zinc, and from .005 to 13.9 mg/l for copper (U.C. Berkeley Mining Waste Study Team, 1988).

Cadmium is extremely toxic to rainbow trout (Ball, 1967). At temperatures between 11.0 and 12.5°C at least 50% mortality occurred at concentrations between .01 and 1.0 mg Cd/l. Concentrations between .008 and .01 mg Cd/l were determined to be lethal to rainbow trout after 7 days. Although the EPA drinking water standard for Cd is .01 mg/l (Lehr et al., 1984), Eisler (1985a) also cited lethal and sublethal effects on freshwater aquatic life at much lower concentrations. .0008 to .0099 mg/l of Cd was lethal to several species of aquatic insects, crustaceans, and teleosts. .0007 to .005 mg/l caused sublethal effects such as decreased growth, inhibited reproduction, and population alterations. When Cd concentration exceeds .003 mg/l in freshwater, adverse effects on fish or wildlife are either pronounced or probable (Eisler, 1985a). Adverse effects are most pronounced in waters of low alkalinity. Brook trout suffered reduction in growth, survival and fecundity in water of low alkalinity with Cd concentrations between .001 mg/l and .003 mg/l. With increasing alkalinity the maximum allowable Cd concentrations increased to between .007 mg/l and .012 mg/l.

Selenium concentrations between .06 and .6 mg/l caused sensitive species of aquatic organisms to die (Eisler, 1985b). Freshwater algae species may fare better if sulfate is present, because for them sulfate has a protective role against Se toxicity. The U.S. EPA drinking water standard for selenium is .01 mg/l (Lehr et al., 1984).

The freshwater organisms that were most sensitive to chromium(VI) are crustaceans and rotifers (Eisler, 1986). Reduced growth, inhibited reproduction, and other adverse effects were found at .01 mg/l  $\text{Cr}^{6+}$  and at .03 mg/l  $\text{Cr}^{3+}$ . The U.S. EPA drinking water standard for chromium is .05 mg/l  $\text{Cr}^{6+}$  and 170 mg/l  $\text{Cr}^{3+}$  (Lehr et al., 1984 and Eisler, 1986).

Lethal concentrations of mercury for aquatic organisms range from .0001 to .002 mg/l (Eisler, 1987). .002 mg/l is the U.S. EPA drinking water standard (Lehr et al., 1984). Eisler also reported significant adverse sublethal effects in aquatic birds at .00003 to .0001 mg Hg/l. The U.S. EPA 1985 criteria for protection of freshwater aquatic life calls for a maximum 4-day average of .000012 mg/l, not to exceed an hourly average of .0024 mg/l (Eisler, 1987).

Eisler claims that these criteria offer only limited protection to aquatic organisms. One of the reasons mercury has high toxicity is that its low toxicity forms can be transformed into forms of very high toxicity through biological processes such as methylation. Mercury also is bioconcentrated in organisms and biomagnified through food chains.

Arsenic is bioconcentrated but not biomagnified in the food chain (Eisler, 1988a). In contrast to mercury, methylation of arsenic greatly reduces its toxicity. The aquatic chemistry of arsenic has been discussed previously in this paper. Important points to remember are that inorganic As is more toxic than organic As and that  $\text{As}^{3+}$  is more toxic than  $\text{As}^{5+}$ . Both inorganic As and  $\text{As}^{3+}$  are also the most mobile forms. Eisler reported that sensitive aquatic species were damaged at .019 to .048 mg As/l. The U.S. EPA 1985 drinking water standard is .05 mg/l (Eisler, 1988a). The 1985 EPA criteria for protection of freshwater aquatic life is .19 mg  $\text{As}^{3+}$ /l. Eisler points out that the effects of chronic low exposure on reproduction, genetic makeup, adaptation, disease resistance, growth, etc. have not been, but need to be, studied.

Adverse effects of lead on aquatic biota were found between .001 and .0051 mg/l. Daphnids were the most sensitive organisms, showing adverse effects on reproduction at .001 mg/l (Eisler, 1988b). The U.S. EPA drinking water standard for lead is .05 mg/l (Lehr et al., 1984). Organic lead compounds are more toxic than inorganic. In water, lead is most soluble and most bioavailable at low pH, low organic content, low suspended sediment, and low salts of Ca, Fe, Mn, Zn, and Cd.

Aquatic organisms are resistant to molybdenum salts -- adverse effects on growth and survival usually only occur at greater than 50 mg Mo/l (Eisler, 1989). Mo is bioconcentrated by selected species of algae and invertebrates but the effect of bioconcentration on higher trophic level organisms is not known. Freshwater fishes are extremely resistant to Mo, but 50% of fertilized eggs of rainbow trout died in 28 days at only .79 mg Mo/l. There are currently no federal drinking water standards for molybdenum. Eisler (1989) proposed criteria of less than .05 mg Mo/l.

Borax ( $\text{Na}_2\text{B}_4\text{O}_5(\text{OH})_4 \cdot \text{H}_2\text{O}$ ) forms during evaporation of enclosed lakes and as an efflorescent mineral on the land surface in arid regions (Hurlbut and Klein, 1977). Boron compounds tend to accumulate in aquatic ecosystems because they are highly soluble. The

U.S. EPA boron criteria for protection of aquatic life is .55 mg/l (Eisler, 1990). Eisler (1990) reported that concentrations greater than .1 mg B/l may affect reproduction in rainbow trout and greater than .2 mg/l may impair survival of other fish species, but additional data is needed. Boron compounds are more toxic to embryos and larvae than adults.

This discussion has mentioned some but not all of the trace metals that may occur in precious metal pit water in toxic proportions. As mentioned, metal toxicity levels vary depending on species as well as on time of exposure, temperature, oxygenation of the water, pH, salinity, hardness, water velocity and interactions between combinations of metals. The U.S. EPA Drinking Water Standards are often used as the basis to determine pit water quality. Although pit water will rarely be used as drinking water, this discussion has shown that the metal toxicity levels for various aquatic organisms are often below the drinking water standard concentrations. If the U.S. EPA Drinking Water Standards continue as the basis for comparison for pit water, aquatic ecosystems will not be protected. Although not discussed in detail here, the effects of pit water quality on waterfowl also need to be seriously considered. It will be impractical, if not impossible, to use netting and/or audio hazing techniques to keep birds out of the pit lakes after mine decommissioning.

## Reclamation

Various methods have been proposed for remediation of the water quality in the Berkeley Pit (Davis and Ashenberg, 1989; Huang and Tahija, 1990). Those methods involve in situ neutralization. Other pits have been reclaimed by pumping the water to a treatment plant. The reclamation phase of metal mining in sulfidic areas may be the most difficult phase in which to control potentially detrimental effects on the environment, and pit water in such an area might have to be treated in perpetuity in order to avoid contamination of groundwater and surface water (Bell and Nancarrow, 1974).

The Rum Jungle uranium mine in the Northern Territory, Australia had three acidic water-filled pits (Dyson's, White's and Intermediate) that have since been reclaimed using three different methods (Harries and Ritchie, 1988). Dyson's open pit had tailings placed at the bottom of the pit. The top surface of the tailings was sloped to discharge water and then covered with a geotextile fabric and a "rock blanket" 1 m thick to carry away groundwater and seepage water. Material from the heap leach and the most contaminated subsoils were placed on top of the rock blanket (above the water table). This was covered with a low permeability geotextile sealing layer, a moisture retention layer, and another sealing layer on top.

Water in both the White's and Intermediate open pits was treated by hydroxide precipitation to raise the pH and remove trace metals (in situ neutralization). The water in White's open pit was then pumped through a treatment plant and returned to the open pit where stratification occurred due to separation of the treated and untreated water. Further treatment of the Intermediate open pit was in situ with the addition of lime, and aeration to ensure mixing. Then the water was allowed to settle and metal hydroxide precipitates were pumped from the bottom of the open pit to the treatment plant. Treatment sludge from the treatment plant had to be buried. Higgs (1990) discussed the chemical stability and disposal considerations of treatment sludge. In general, sludges consist of metal hydroxides and gypsum. They are not always stable during storage and constituents may redissolve. If sludges will be produced by a treatment process, their storage or disposal is an important consideration.

The ideal situation for reclamation of pit lakes would be to be able to predict pit water quality so well that upon closure, but before the pit begins filling, plans could be made to protect the future water quality. This might include covering reactive materials, or adding a layer of neutralizing material to the pit bottom. If arsenic is anticipated to be a problem, the pH may need to be lowered with the knowledge that other elements may then become a problem. More ideas are needed in this new area and each mine will need a unique solution. Planning ahead could prevent an expensive reclamation problem.



## Conclusion

Many of the open pit precious metal mines that are scattered throughout Western North America, with a high concentration in Nevada, will ultimately become pit lakes. The lakes may be meromictic if total dissolved solids concentrations are high enough. The lakes may be stratified or mixing or have periods of both depending on the local climate and on morphometry of the pits.

It is important to know the regional groundwater flow system prior to mining since flow directions and water table elevations have direct bearing on the characteristics of the pit lake. Water may be expected to seep from the pits during periods of high rainfall or surface water runoff, and to flow back into pits through aquifers during periods of high evaporation. Subsidence can occur within the cone of depression during dewatering, leading to an altered groundwater flow regime when dewatering ends.

Reactions of water with exposed wallrock in an open pit mine are major contributors to pit water quality. Flow in joints and fractures makes calculation of exposed surface area difficult. Sloughing of wallrock occurs, adding new surface area for reaction.

The pH of pit water is determined by the mineralogy of the surrounding rocks. Sulfide minerals, when exposed to oxygen and water, oxidize in a series of reactions that produce acid. Bacteria can greatly accelerate the rate of the oxidation reactions. Dissolution of minerals such as calcite can neutralize acid. The balance between acid producing minerals and acid consuming minerals is one measure of the potential for acid production. Other tests consider rates of oxidation and neutralization reactions. All of the tests have limitations that need to be recognized before they are used to make predictions. Control of acid generation and migration and/or treatment of acid can be difficult undertakings that need to be planned for well in advance.

Trace element concentrations are often high in waters associated with mining. Most trace elements are soluble at low pH, but adsorb onto iron oxyhydroxides which form at high pH. Arsenic is a trace element that naturally occurs in high concentrations in groundwater in the western United States. In pit water at or close to neutral pH's, arsenic will generally be soluble under anoxic conditions and will generally adsorb or coprecipitate with iron under

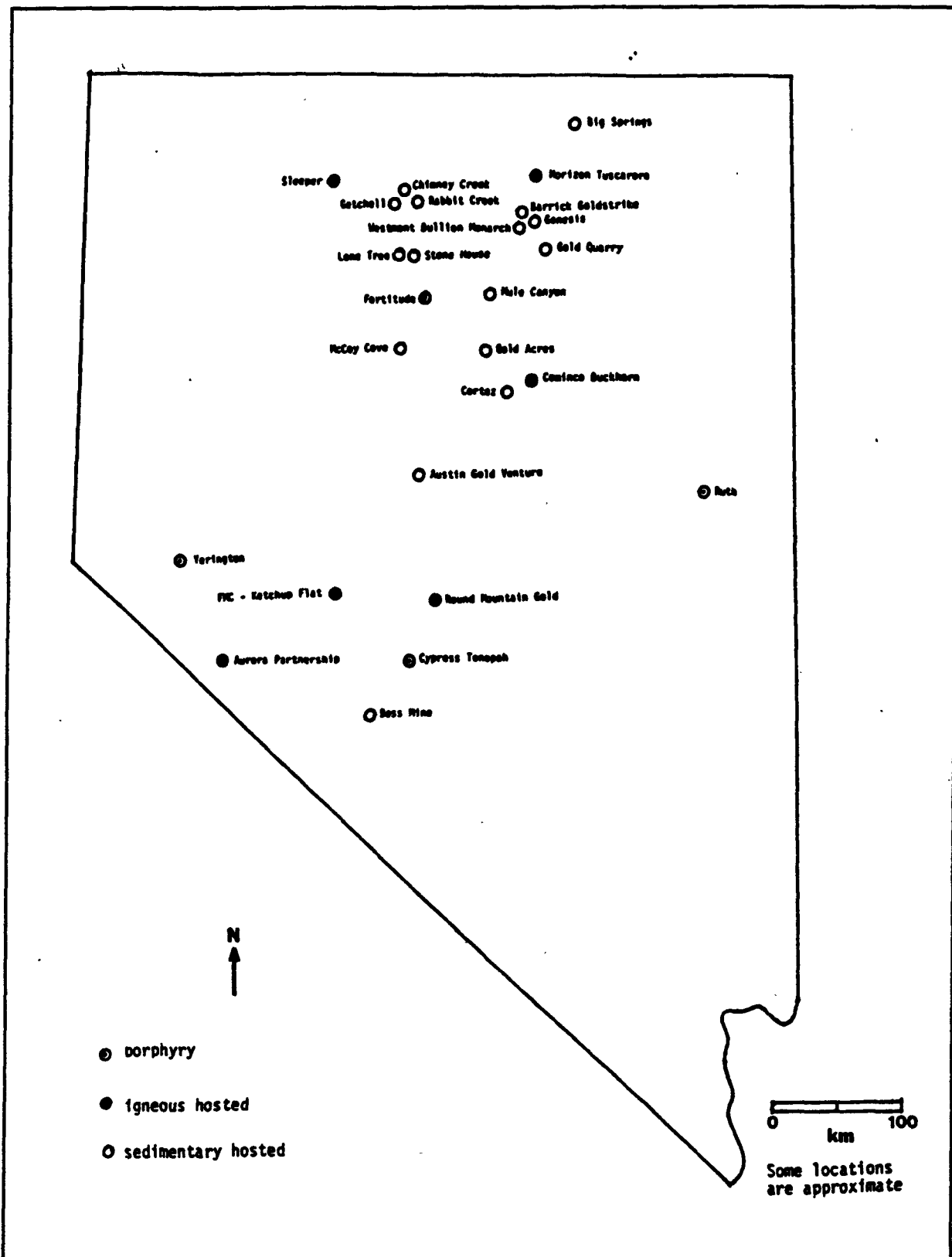
oxic conditions. Optimum removal of arsenic species from solution occurs at pH values of 4 to 5 with excess iron concentration relative to arsenic.

Because of the arid climate in which most precious metal open pit mines are located, evapoconcentration will increase the concentration of trace elements in pit water. Hydrothermal input to pit lakes will affect their temperature, stratification, circulation and chemistry.

Data from existing water-filled pits from phosphate, uranium, coal, copper, silver and gold mining yields valuable information about conditions that can be expected in pit lakes. However, more limnological studies are needed in these existing pit lakes, to collect data versus depth at different times of the year to determine seasonal patterns. All pit water data collected should include a statement of how the samples were collected, stored and analyzed including ionic balances so adequate modeling can be done.

Effects of pit water quality on aquatic biota, birds and humans need to be considered. Water quality standards that protect wildlife are generally lower than the drinking water standards, making the latter an inadequate predictor of the ultimate quality of the water for the survival of aquatic biota. Reclamation of pit water can be difficult, unless poor water quality was anticipated and preventive measures were undertaken.

Figure 17 shows the geology of Nevada's future and current pit lakes. By looking at data from existing pit lakes, we saw that there is enough variability in pit water quality among the porphyry copper deposits -- Butte (Montana), Ruth and Yerington -- to show that gross generalizations about geology are not enough to predict pit water quality. It may be safe to assume, however, that the sedimentary hosted deposits generally have calcite associated with them, and have lower sulfide content than the porphyries, so acid may not be a problem. Arsenic concentrations in pit water will often be high due to abundant naturally occurring arsenic in Nevada. Each pit is a unique system and should be studied as such.



**Figure 17**

## References

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