Chapter 13

ACID MINE DRAINAGE AND OTHER MINING-INFLUENCED WATERS (MIW)

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13.1 INTRODUCTION

by R. L. Schmiermund

The origins of the phrase "Acid Mine Drainage" (AMD) are obscure but it was probably first used to describe the low-pH effluent from certain underground mine workings. In view of its common use today as a descriptive phrase applied to most types of mining-influenced waters, the term is inadequate due to its limited scope and consequently is often misleading. The more recent modification, "acid rock drainage" (ARD), de-emphasizes the obvious genetic relationship of mining to the vast majority of such waters but is unfortunately no more inclusive than its predecessor.

The objective of this chapter is to provide an overview of the characteristics of natural waters affected by mining activities (exclusive of direct contributions and discharges from milling and other beneficiation operations) and to briefly discuss the geochemical processes relevant to those characteristics. Such waters include groundwaters that enter underground workings and exit via surface openings or are pumped to the surface, groundwaters that enter pits or surface excavations and contribute to surface water courses, meteoric precipitation which, in the course of becoming runoff or recharge to surface water bodies, contacts pit faces, ore and waste-rock piles or tailings piles. Many such waters are indeed acidic with pHs significantly less than 7.0, thus giving legitimacy to the term "AMD." The fact that all these generic rock-water interactions may also take place, with similar impacts to water quality, in the absence of mining validates the term "ARD". Indeed, situations occur where ore bodies naturally outcrop at the surface and are actively weathering and impacting surface water courses. One such example is the Vangorda base-metal massive sulfide orebody near Faro, Yukon Territories. Others are described by Runnells et al. (1992) and Plumlee et al. (1993). These situations are rare compared to those where mining has been directly responsible for modifications of water quality.

Low pH is not a universal characteristic of natural waters influenced by mining. Drainage from mines may also be neutral or basic, and the most environmentally serious aspect of some drainage may be not be its low pH, but rather its heavy metal content, iron or aluminum content, sulfate or arsenate content or suspended solids. To give the impression that low pH is a universal characteristic of mining-related waters, or that low pH is the only characteristic of concern, is misleading to the general public and may impair the ability of the mining industry to evaluate situations and plan for economically viable mining operations. Taken together, these points argue for a more accurate term such as "mining-influenced-water(s)" (MIW(s)). Such a term invites a definition of the nature and extent of the influence rather than simply declaring, in an automatically negative and perhaps erroneous way, that the characteristic of greatest concern is low pH. While the inconvenience of a new term and acronym is acknowledged, the benefits appear sufficient to justify its adoption and use.

Figure 1 schematically illustrates five important and generally undesirable characteristics of MIWs and the relationships of AMD and ARD. Any given MIW may possess none, one or more or all of these characteristics - thus they should be regarded as potential characteristics for any given MIW. While certain characteristics are unquestionably genetically related, the geochemical processes that give rise to and/or tend to mitigate each characteristic will be discussed separately. Each characteristic requires a different approach to prediction, monitoring and control. MIW issues are common to virtually every type of mining operation, regardless of mining method, commodity or type of ore body, although some deposits, mine locations and climates result in minimal concerns.

A significant literature is available on the subject of MIW (typically referred to as "acid mine drainage") but a great portion is disseminated among an unusually large number of journals, publications of various government agencies and difficult-to- obtain conference proceedings. A recent and ongoing focus of the U.S. Geological

Survey has been to establish guidelines for the prediction of MIW characteristics of new mining operations based on ore-body types (Smith (1994), Plumlee et al. (1993), Ficklin et al. (1992)). Current activities of the U.S.G.S. Mine Drainage Interest Group may be found via http://water.wr.usgs.gov/mine/home.html. Gray et al. (1994) provides additional evidence of efforts to place the prediction of MIW quality in the context of economic geology evaluations of mining properties. A cooperative of various Canadian government agencies and mining companies has established Mining Environment Neutral Drainage (MEND) to coordinate a large volume of research on MIW situations in that country (http://www.emr.ca/mets/mend/). A particularly useful overview of MIW issues is presented by the British Columbia Acid Mine Drainage Task Force (BCAMDTF, 1989).

13.2 POTENTIAL CHARACTERISTICS OF MINING-INFLUENCED-WATERS

13.2.1 GENERAL

Natural surface and subsurface waters may be affected by mining operations in a variety of ways, leading to a range of chemical and physical characteristics. A thorough knowledge of these characteristics is critical to cost-effective environmental compliance and management and not infrequently has significant impacts on mine planning. Investigations of existing MIW situations should initially seek to define and quantify those characteristics, without preconceptions and as completely as possible, both in absolute terms and relative to an applicable background. Similarly, spatial and temporal variations in chemical and physical characteristics should be established. Investigations such as baseline studies and environmental impact analyses for planned mining operations, which frequently include existing MIW situations, are poorly served by focusing only on regulated constituents rather than seeking to fully characterize waters. Not only do regulations change over time, but requirements for understanding water systems in the context of treatment or utilization typically escalate as a project goes forward. The cost of 'doing it right the first time' is usually trivial compared to the cost of dealing with incomplete information about characteristics and the resultant indecision or the possibility of incorrect decisions at a later time.

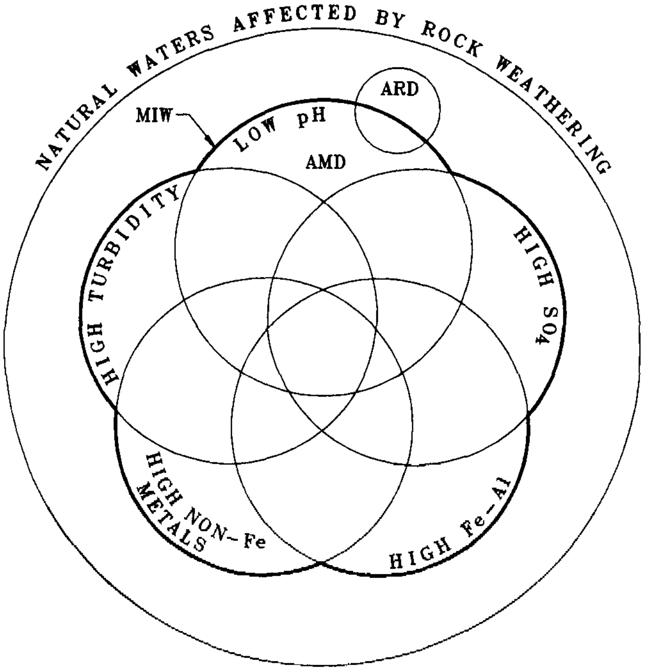

Figure 1a Venn diagram illustrating the five potential characteristics of MIW discussed in this chapter and the relationships of AMD and ARD to MIW. Note that MIW is a subset of all waters affected by rock and mineral weathering and that AMD is a subset of MIW. ARD includes AMD and acidic non-MIW waters.

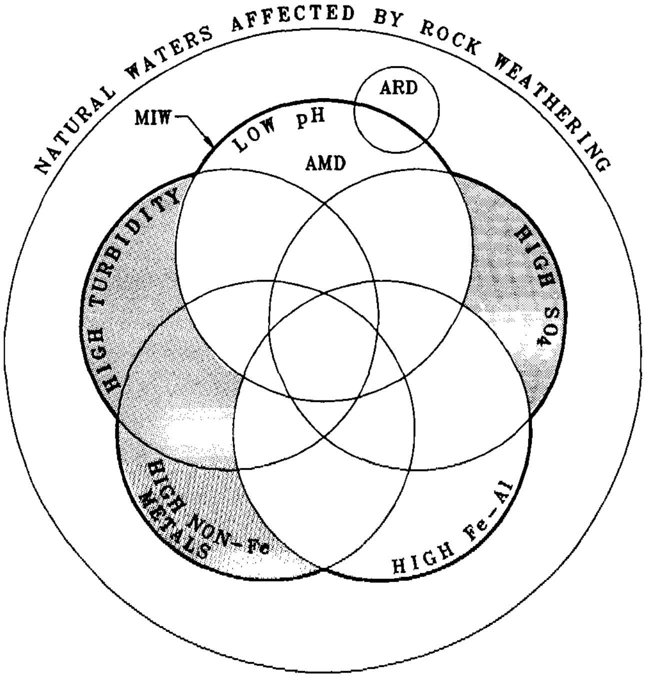

Figure 1b Example illustrating a non-acidic MIW with high sulfate, turbidity and elevated heavy metals (e.g., neutralized water with suspended ferric oxyhydroxide and elevated copper, pH = 6, Eh = +800 mV, Cu = 6 mg/L, Fe < 0.001 mg/L).

Anticipating potential effects of a mining operation on natural water quality characteristics is considerably more difficult than characterizing existing conditions. The topic is not discussed here but the reader is referred to EPA (1994) for a survey of general materials-testing methods and predictive models and Scharer et al. (1994) for a review of models applied to tailings. Documented characteristics of local MIW can provide valuable guidance for prediction of future conditions. If local MIW situations are present, are thoroughly understood and can be logically shown to be relatable in certain ways to proposed operations, it may be possible to predict something useful - even if it is only the worst-case scenario. Literature searches for MIW cases related to similar deposits or analogous unit operations may also produce valuable clues but in the end, site-specific knowledge is required because of the many factors involved in determining the characteristics of MIW.

Table 1 presents an assortment of selected MIW analyses which serve as illustrations of the characteristics discussed here. At first glance it is obvious that there is little consistency in the type of geochemical data collected or reported. In general, incomplete or partial analyses and incomplete documentation appear to be the rule rather than the exception.

13.2.2 FIVE COMMON CHARACTERISTICS OF MIW

The MIW characteristics that are most consistently of concern are elevated pH and heavy metal concentrations, thus it is useful to consider their ranges. Ficklin et al. (1992) and Plumlee et al. (1993) discuss and classify MIW from some 25 identified and geologically classified metal mines in the Colorado Mineral Belt and include six MIW-like waters related to un-mined metal deposits. Their MIW classification is based on pH and the content of combined non-ferrous metals. As shown in Figure 2, pH varies from less than zero to slightly greater than 8.0 and combined dissolved metals (Zn+Cu+Cd+Co+Ni+Pb) range over five orders of magnitude. This pH range is not inclusive of all MIW given the observations of -3.0 pH values at Iron Mountain, California (Alpers and Nordstrom, 1991). Smith et al. (1994) show that the concentration variations for individual metals may be even more pronounced (seven orders of magnitude) than is the case for combined metals. Certain deposit types appear to have consistent MIW characteristics while MIWs for other types vary widely. The data clearly supports the fact that not all MIW is acidic and that unacceptably high metal concentrations are not necessarily associated with low pH.

Although pH and concentrations of heavy metals as discussed above are clearly important, and often the most important, characteristics of MIW, the analyses listed in Table 1 illustrate the preponderance of other components in most MIW, some of which (eg. $SO_4^=$, Fe, and Al) are subject to compliance criteria in some cases. Dissolved iron and aluminum typically occur in significantly higher (10 times to 100 times) concentrations than the heavy metals mentioned above (Smith et al., 1994). These two metals are worthy of separate consideration as a characteristic of MIW, not merely because of their higher concentrations, but also because they have distinct geochemical characteristics. Frequently iron and aluminum in MIWs cause pronounced visual effects such as the formation of "yellow-boy" and, in some cases, influence the behavior of heavy metals. Other metals such as the alkali earths Ca and Mg, and the alkali metals Na and K may also occur in greater concentrations in MIWs but are typically not of environmental concern *per se* but may limit some uses of MIW and in some cases are tied to toxicity indices for other metals (e.g., Zn toxicity as a function of hardness).

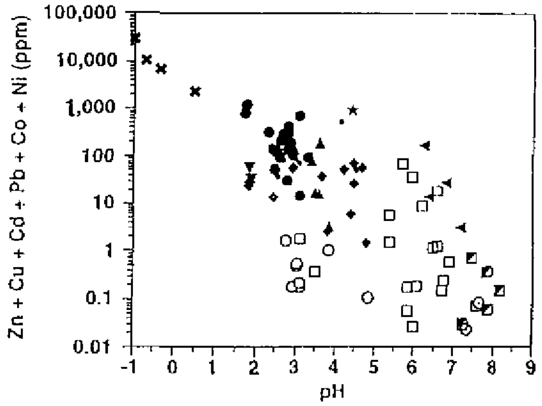

Figure 2 Variations in pH and metal content of Colorado Mineral Belt MIW. Different symbols represent different genetic types of mineral deposits with different mineral assemblages, host rocks or alterations. (after Smith *et al.*, 1994).

Anion concentrations also represent important characteristics of MIWs and, as is the case of $SO_4^=$ and As (which occurs as $H_2AsO_4^-$ or $HAsO_4^=$), some have associated compliance criteria. Since sulfur, as sulfide, is associated with the majority of the mineral phases that represent the sources of metals in MIWs, and since relatively few natural processes remove sulfate from surface or ground waters (i.e., $SO_4^=$ behaves conservatively), it is logical that its concentrations would typically exceed those of metals (X000 mg/L are not uncommon). This conservative aspect of sulfate geochemistry also dictates that elevated sulfate is typically the most pervasive and persistent characteristic

Table 1 Chemical analyses of selected mining-influenced waters

	SOURCE	1	2	3	4	5	6	7	8	9	10	11	12
pH (field)	units	6.8	2.96	7.84	-1	1.8	4.4	7.7	6.2	3	3.2	8.15	2.65
Acidity	mg/L as H2S									342			305
Eh (field)	mv	280	500	250									840
DO	mg/L						7.5	7.5			10		
Condtivity	uS	<100	2100	1170		38000	980	2800		1780		3036	1274
TDS	mg/L		2568	1220						1260			
TSS	mg/L		0.6	419.6								54	
Temp	C	5	13	14	42							10	6
Sulfate	mg/L	19	1520	720	760000	130000	640	1200	1800	817	2000	1400	950
Chloride	mg/L	2	0.5	1.4						22		16.2	
Alkalinity	mg/L as CaC	27.2	<1	73		n.d.	n.d.	330		<1	n.d.	401.4	<1
Fluoride	mg/L		0.28							0.1		1.08	
Ca (Tot/Dis)	mg/L	/9.8	96.6/	280/249	/279					119/	/400	180/	/36
Mg (Tot/Dis)	mg/L	/2.5	6.73/	23.8/23.9	/437					68/	/110	122.5/	/26
Na (Tot/Dis)	mg/L	/2.4	3.6/	2.6/2.1	/416					17/	/19	410/	/0.79
K (Tot/Dis)	mg/L		1.68/	2.14/1.79	/194						/4	26.5/	/2.4
Si (Tot/Dis)	mg/L	/6.11			/14					10/		6.4/	/5.6
Fe (Tot/Dis)	mg/L	/<0.02	63.16/	22.4/0.07	/111000	/28000	/2.4	/0.02	/87	15/	/37	0.13/	/27
Al (Tot/Dis)	mg/L	/<0.05	43.71/	0.66/<0.05	/1420	/5400	/2.6	/<0.1	/0.2	29/			/19
Ag (Tot/Dis)	mg/L		0.003/	0.004/0.00	/2.4								
As (Tot/Dis)	mg/L	/<0.001	0.039/	0.156/<0.0	/340	/15					/0.12	<0.005/	
Cd (Tot/Dis)	mg/L	/0.0002	0.422/	0.054/0.02	/211	/3	/0.29	/0.001	/0.12			0.0002/	
Co (Tot/Dis)	mg/L				/5.3	/16	/0.02	/0.02	/0.12			0.004/	
Cu (Tot/Dis)	mg/L	4.95/3.2	4.6/	0.11/0.04	/4760	/460	/2.4	/0.001	/0.27		/5.4	0.011/	
Pb (Tot/Dis)	mg/L	/<0.02	0.04/	2.71/0.02	/11.9	/<0.2	/0.01	/<0.005	/0.15			0.0115/	
Mn (Tot/Dis)	mg/L	/0.01	36.84/	40.9/30.4	/22.9					10/	/31	0.05/	/5.3
Ni (Tot/Dis)	mg/L		0.13/	<0.01/<0.0	/3.7	/21	/0.04	/0.02	/0.19				
Zn (Tot/Dis)	mg/L	/0.04	72.77/	14.5/8.87	/23500	/390	/69	/0.11	/170		/9.6	0.042/	

n.d. = not detected   xxx/yyy = total (unfiltered) concentration / dissolved (filtered) concentration

SOURCES:

1 - Chalcopyrite massive sulfide, preCambrian meta sediments, Osceola Tunnel drainage, 0.45 um, Ferris Haggarty Drainage Study, Adrian Brown Associates 321A/94083

2 - Acid-sulfate altered Au-bearing stockwork, northern Peru, tunnel drainage, undisclosed data source

3 - Limestone-hosted Pd/Zn massive sulfide manto, northern Peru, tunnel drainage, 0.45 um, undisclosed data source (pH at ore face <3)

4- Richmond Mine, Iron Mountain, CA , tunnel drainage, 0.1 um, Alpers and Nordstrom (1991)

5 - Acid-sulfate epithermal Au deposit, Blackstrap Mine, Summitville, CO, dump drainage, 0.1 um, Plumlee et al. (1993)

6 - Limestone hosted Pb-Zn-Ag manto deposits, Yak Tunnel, Leadville, CO, tunnel drainage, 0.1 um, Plumlee et al. (1993)

7 - Gold-Telluride vein deposit, Carlton Tunnel, Cripple Creek, CO, tunnel drainage, 0.1 um, Plumlee et al. (1993)

8 - High-sulfide carbonate hosted, Wellington Mine, Breckenridge, CO, tunnel drainage, 0.1 um, Plumlee et al. (1993)

9 - Anthracite coal mine-affected creek, Shamokin Creek at Weighscale, PA , presumed unfiltered, Hem (1985)

10 - Quartz-sulfide Au veins in metasediments, Argo Tunnel, tunnel drainage, 0.45 um, Idaho Springs, CO, Stewart and Severson, 1994

11 - Bituminous coal strip-mine effluent after settling pond, Zowada Pit, Big Horn Mine, Sheridan Co., WY, presumed unfiltered, Dettmann **et al. (1976)**

12 - Bituminous coal underground mine effluent, Babb Ck. watershed near Morris, PA, 0.45 um, Crouse and Rose (1976)

of MIWs (Webster et al.(1994) provides an example of sulfate persistence). Arsenic, although occurring in ore deposits as mineral arsenides and sulfides (sometimes abundantly) and is most commonly present as an oxyanion in solution (Welch et al. (1988) and Hem (1985), it is typically not as concentrated in MIW as sulfate and does not behave conservatively as does sulfate. Dissolved arsenic concentrations, in general, are similar to metal concentrations.

A final important characteristic of MIW is total suspended matter and resultant turbidity. This characteristic is highly variable, both spatially and temporally, and a strong function of the geochemistry of iron and, to a lesser extent, aluminum. Significant concentrations of iron are possible under low-pH conditions, but iron may also be transported in MIW at higher pHs as suspended ferric oxyhydroxide (FeOx) solids. Low-pH MIW is typically very clear because all constituents, including iron, are in solution. Progressive neutralization of pH and/or oxidation of MIW causes amorphous FeOx to precipitate as initially sub-micron sized material that remains in suspension, thus causing turbidity. As the precipitated particles continue to grow and aggregate they eventually reach a critical mass and settle out of the water column, thus reducing turbidity. Turbidity is an important characteristic of MIW, not only because it is regulated in some cases, but also because FeOx precipitate has a large surface area and a well documented facility for adsorption of metals and arsenate. Thus, when FeOx suspensions are present, some portion of otherwise dissolved components of MIW may be associated with suspended matter and not detected in filtered water samples. When FeOx suspensions eventually leave the water column, metals may go with them and form part of the temporarily immobile substrate of streams that can be remobilized under different flow conditions.

13.3 GEOCHEMICAL PROCESSES RELATED TO THE CHARACTERISTICS OF MINING-INFLUENCED-WATERS

Each of the characteristics of MIW discussed in Section 2 may be related to one or more geochemical processes that either give rise to or attenuate a characteristic feature. Although there is considerable overlap in attributing MIW characteristics to geochemical processes (e.g., one process may be responsible for multiple characteristics), each characteristic is unique in one or more ways so that the presence or absence of any one characteristic feature is not a guarantee of the presence or absence of any other characteristic feature. In short, the evolution of MIW is a complex site-specific function of parent materials and prevailing conditions.

13.3.1 pH, ACIDITY AND ALKALINITY CONTROLS

13.3.1.1 Definitions and Analytical Issues

The terms pH, acidity and alkalinity are very commonly used in relation to MIW issues and merit a brief discussion. Understanding the analytical aspects of these and all chemical quantities are extremely important and the burden of QA/QC should never be assumed to lie entirely with the analytical laboratory.

The hydrogen ion activity ($[H^+]$), expressed as pH (pH = $-\log[H^+]$), is a straight-forward *intensity* attribute of all aqueous solutions but may be incompletely understood. For example, the commonly used pH-scale ranges from 0-14 pH units, but these limits are only an artifact of common use. Although uncommon, waters in nature may have pH values outside this range: acidic MIW is one situation where this occurs. Neutral pH (pH ≈ 7.00) is merely the point at which $[H^+] = [OH^-]$ and should not automatically be assumed to be the "background" pH of a natural water as most natural water have a pH between 6 and 9. Waters are commonly referred to as being "acidic" or "basic" when their pH is below or above neutral pH, respectively, but "acidic" in this context should not be confused with "acidity".

Acidity and alkalinity are *capacity* attributes of aqueous solutions and neither are exclusively related to pH. Acidity is the capacity of a solution to consume hydroxide ions and any reaction or component (including proton) that accomplishes this is a contributor to acidity as illustrated by the following Equations. High acidity represents a resistance toward neutralization and a facility for corrosion.

$$OH^+ + H^- \rightarrow H_2O \qquad (13.3.1.1\text{-}1)$$

Alkalinity is the capacity of a solution to consume protons and may also be illustrated by Equation 13.3.1.1-1 but most often interpreted in terms of the inorganic carbon reactions shown in Equation 13.3.1.1-3. However in metal-rich MIW, reactions of the type illustrated by Equation 13.3.1.1-4 may be the predominant sources of alkalinity.

$$\begin{aligned} &CO_3^= + H^+ \rightarrow HCO_3^- \\ &HCO_3^- + H^+ \rightarrow H_2CO_3^0 \end{aligned} \qquad (13.3.1.1\text{-}3)$$

$$Al(OH)_n^{(3-n)} + nH^+ \rightarrow Al^{3+} + nH_2O \qquad (13.3.1.1\text{-}4)$$

It is important to recall that neither acidity nor alkalinity are absolute quantities as are $[H^+]$, pH or concentrations of a metal or sulfate. They are strictly operationally defined by the analytical procedure - in effect they represent bench tests conducted under specified conditions. In the case of alkalinity, the typical reporting units (mg/L as $CaCO_3$) are based on the assumption that inorganic carbon is the only source of alkalinity, which, as stated above may well not be the case for MIW. It is preferable to request the analytical laboratory to also report alkalinity in terms of the raw experimental data (e.g., x ml of acid of N normality was used to obtain a specified endpoint pH). Such data may then be used by the geochemist to correctly interpret the characteristics of MIW. Refer to Standard Methods for the Analysis of Water and Wastewater (Greenberg et al., 1992) for these and other analytical procedures.

13.3.1.2 Sulfide Mineral Oxidation and Production of Acidity

Sulfide minerals, especially the most common of all sulfide minerals, pyrite (FeS_2), are at the heart of most MIW problems. Sulfide minerals represent the form in which most base metals occur in nature, iron-sulfide minerals (pyrite, pyrrhotite, marcasite) and other sulfides are common associates of important non-sulfide ore minerals (e.g., native gold), and iron-sulfides are commonly associated with coal and the rock sequences that host coal.

Pyrite, marcasite and pyrrohotite form and are stable under chemically reducing, typically anoxic conditions and consequently are inherently and thermodynamically unstable in the presence of the earth's atmosphere and surface environment. It is pyrite's response to this instability that gives rise to elevated concentrations of protons, iron and sulfate in many MIWs and initiates processes that produce other characteristics. In the overall sense, the reactions related to this instability are referred to as "oxidation" although the process is far more complex than simple oxidation. The wide range of rates at which pyrite succumbs to this inherent instability, and the intricate relationship of those rates to environmental variables attests to the complexity of the overall process. There is a vast literature on the subject of pyrite oxidation which can only be alluded to here. The reader is referred to and Alpers and Blowes (1994) for a recent collection of relevant articles, and to Evangelou and Zhang (1995), de Hann (1991), Lowson (1982), Nordstrom et al. (1979), Rogowski et al. (1977), Singer and Stumm (1970) and Barnes and Romberger (1968) for reviews of and a perspective on the subject.

Processes associated with the oxidation of pyrite are summarized in Figure 3 and have been divided into initiation and propagation steps. The initiation step requires molecular oxygen in a aqueous or humid (Borek, 1994) environment and apparently can proceed by two different paths including dissolution of pyrite with subsequent microbial-assisted oxidation of sulfide sulfur or by direct oxidation of pyrite to sulfate (Stumm and Morgan, 1981). Although both initiation steps ultimately results in aqueous solutions with three of the common characteristics of MIW (increased H^+, $SO_4^=$ and Fe^{2+}) it does not account completely for field observations of stoichiometry or kinetics associated with pyrite weathering. For example, the overall initiation reaction (Equation 5) only produces two moles of protons for every mole of pyrite oxidized and thus does not account for observed acidities and pHs.

$$FeS_{2,py} + \frac{7}{2}O_2 + H_2O \rightarrow Fe^{2+} + 2SO_4^{2-} + 2H^+ \tag{13.3.1.2-5}$$

As an aside, the initiation reaction (Equation 5), although an incomplete description of acidity production during pyrite weathering, is the basis for calculating Acid Producing Potential (APP) from total sulfur determinations (i.e., it is the basis of the 31.25 factor commonly used in such calculations) and for interpreting Acid-Base Accounting results. Thus, Acid-Base Accounting methods do not take into account the all-important propagation phase of pyrite weathering discussed below.

The propagation phase of pyrite weathering involves the oxidation of Fe^{2+} to Fe^{3+} and the subsequent oxidation of additional pyrite by Fe^{3+}. Oxidation of ferrous iron in a sterile (abiotic) environment proceed very slowly under acid conditions and would probably be the overall rate-limiting reaction for pyrite weathering. Acidic mine drainages might never be produced under such conditions. However, certain microorganisms that thrive in dark, acidic environments with very minor amounts of oxygen, derive energy from the oxidation of ferrous iron and thus catalyze the reaction and increase rates of Fe^{3+} production by five or six orders of magnitude over abiotic rates. The organism thought to be chiefly responsible is the acidophilic, obligate chemotroph, Thiobacillus ferrooxidans bacteria. Depending on the prevailing pH environment, Fe^{3+} may either attack additional pyrite (Equation 6, the overall propagation phase reaction) or form ferric oxyhydroxides or jarosite. Both processes liberate additional protons and contribute to acidity in the system.

$$14Fe^{3+} + FeS_2 + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \tag{13.3.1.2-6}$$

13.3.1.3 Neutralization and Acquisition of Alkalinity

Neutralization refers to raising the pH of acidic MIWs and thus altering one of its distinguishing characteristics.

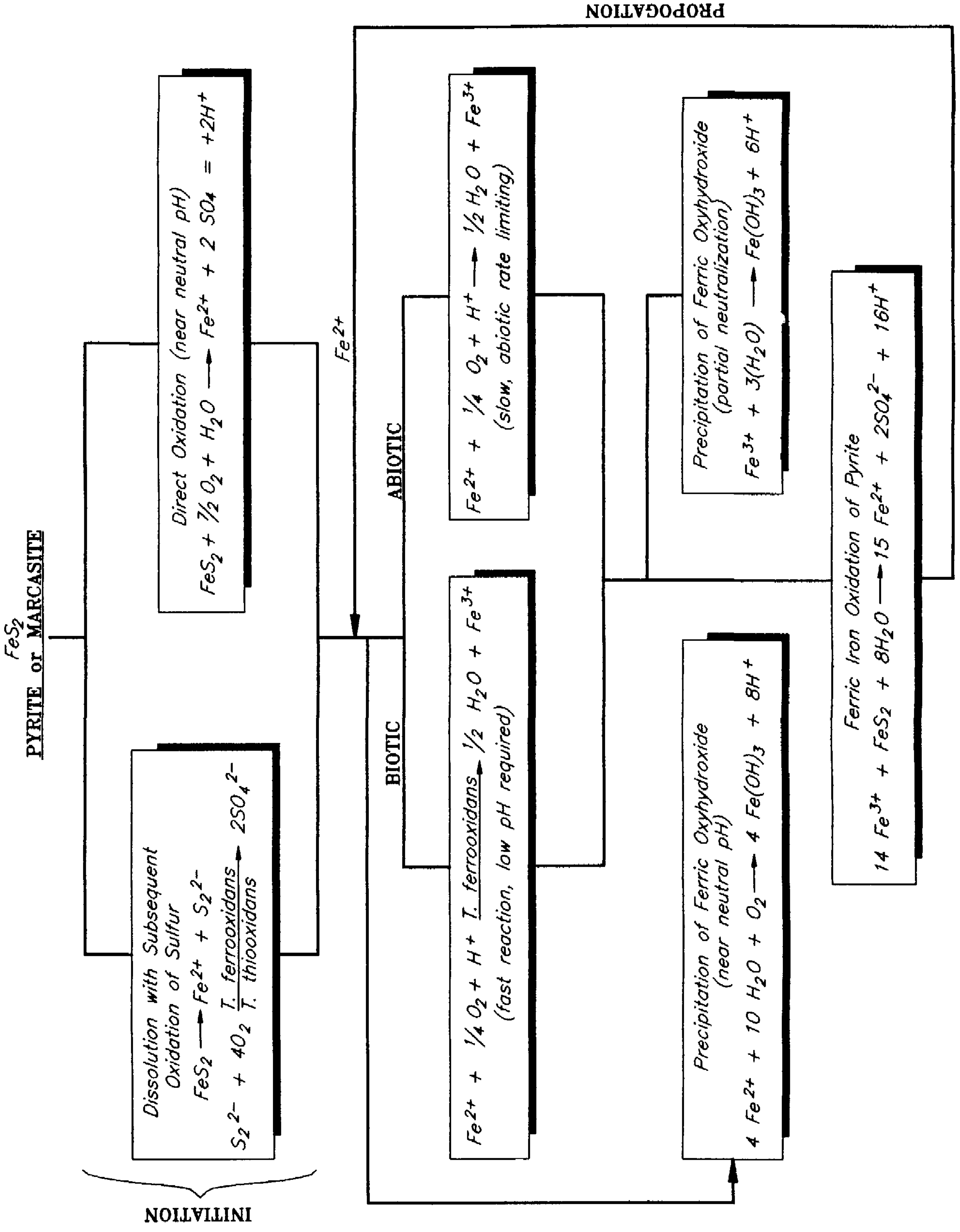

Figure 3 Flow chart of major reactions and pathways involved in weathering of pyrite.

However, other important alterations may occur in concert with raising the pH, depending on the extent of the neutralization.

When acidic MIWs flow away from the environment in which they were created (typically sulfide mineral-rich) they may be exposed to rock types capable of reacting with the aqueous products of sulfide oxidation, especially the protons. The resultant neutralization gives rise to MIWs that are no longer acidic but may still contain dissolved metals. Extensive neutralization, in conjunction with oxidation may also cause metals to precipitate as hydroxide or oxy-hydroxide solids, sometimes causing significant turbidity.

Rocks containing carbonate minerals (e.g., calcite) are typically thought of as agents of neutralization but the process may also occur in the presence of silicates, especially feldspars. Carbonate minerals generally dissolve completely in acidic solutions (congruent dissolution) but feldspars frequently dissolve incongruently resulting in some of the original components being solubilized and the remainder forming a new solid phase such as a clay mineral. Carbonate minerals also react rapidly (relative to silicates) but their effectiveness in neutralization reactions will be controlled by available surface area per unit volume of MIW and may be compromised by "armoring" of the carbonate surfaces. Armoring occurs when low-solubility secondary solid products of neutralization (e.g., ferric oxyhydroxides) coat the carbonate surfaces and limit further reactions. The exact products of incongruent silicate dissolution, and thus the extent of neutralization by silicates, are highly dependent on the chemical environment. Consequently the net effect of MIW contact with silicates changes as the composition of the MIW changes along its flow path. Both congruent and incongruent dissolution reactions result in decreases in solution proton concentrations as well as increases in dissolved components, some of which in-turn can consume protons. Such components constitute alkalinity.

Alkalinity is defined operationally as described above in Section 13.3.1.1 but is also equivalent to the sum of constituents that can consume protons less the protons already present as generally shown in Equation 7. A MIW may acquire components of alkalinity by dissolution of rocks and minerals or by mixing with waters containing alkalinity. The effectiveness of contact with rocks for neutralizing acidic pHs and supplying alkalinity is a function of reaction rates as discussed by Sherlock et al. (1995). Carbonate minerals typically have considerably higher reaction rates and are therefor more effective in neutralizing MIW. The effectiveness of mixing for neutralization on a volume-for-volume basis can be assessed by performing simple titrations of the two subject waters.

$$\begin{aligned}Alkalinity = \\ &-[H^{+}]+[OH^{-}]+[HCO_3^{-}]+2[CO_3^{2-}] \\ &+[NH_3]+[HS^{-}]+2[S^{2-}] \\ &+[HSiO_3^{-}]+2[SiO_3^{2-}]+[B(OH)_4^{-}] \\ &+n[Me(II)(OH)_n^{2-n}]+n[Me(III)(OH)_n^{3-n}]\end{aligned} \tag{13.3.1.3-7}$$

Results of MIW contact with various minerals (i.e., the extent of neutralization) may be predicted through the use of a variety of geochemical computer codes. Simplistic models assume complete reactions of some mass of rock with a unit volume of MIW (a situation analogous to adding lime to a stirred reactor) but are unrealistic predictors of many situations. Similar assumptions are made in Acid-Base Accounting calculations. More comprehensive models would take into account effective surface areas, diffusion rates, reaction rates, boundary layers and other aspects of mass transport. Such models are difficult to apply in real-world cases such as the flow of MIW through a tunnel driven in limestone or meteoric precipitation percolating through a waste rock pile.

13.3.2 SULFATE AND ARSENATE CONCENTRATIONS

An elevated sulfate concentration is perhaps the most consistent characteristic of MIW because the majority of cases involve oxidation of sulfide minerals. Figure 3 show the production of sulfate during the oxidation of sulfide minerals and it is clear that each mole of pyrite must ultimately produce two moles of sulfate. Protons are also produced but, as described in Section 13 3.1.2, neutralization can negate this characteristic. Since pyrite and other sulfides are abundant in most metal mining environments, the concentrations of sulfate in the associated MIW will not be source limited. Rather the concentrations will reflect the effectiveness of sulfide mineral oxidation and the volumetric discharge of water from the site of the oxidation and any subsequent dilution.

The limited number of chemical mechanisms that remove sulfate from MIW solutions is also a reason for the common occurrence of elevated sulfate. Gypsum ($CaSO_4 \cdot 2H_2O$) and secondary iron sulfate minerals like jarosite that can form from the oxidation of pyrite are far too soluble to effectively decrease sulfate concentrations in MIW (Cravotta, 1994). The approximately conservative nature of sulfate in MIW streams provides a basis for assessing and tracing the fate of non-conservative components, especially metals. For example, a downstream decrease in sulfate concentrations can frequently be ascribed to dilution. Other components that decrease to a greater extent in the same reach of stream may be assumed to be attenuated by one mechanism or another (e.g., precipitated along with ferric-oxyhydroxide particles or sorbed onto bed sediments).

Arsenic is included here with sulfate in light of the tendency for arsenic to form complexes with sulfide sulfur and thus to form arsenic sulfides including orpiment (As_2S_3) and realgar (As_2S_2) and metal sulfarsenides such as arsenopyrite (FeAsS). In addition, the oxidized form of arsenic (As(V)) as arsenic acid hydrolyses to form oxyanions (e.g., $H_2AsO_4^-$, $HAsO_4^{-2}$) as does sulfuric acid and there appears to be a limited solid solution series between scorodite ($FeAsO_4 \cdot 2H_2O$) and jarosite. Selenium, antimony and possibly tellurium could be logically included in this characteristic but will not be discussed here.

Oxidative weathering of arsenopyrite is probably the most common source of arsenic in MIW and occurs under less oxidizing conditions than does pyrite oxidation (Marsden and House, 1993). Arsenopyrite is actively oxidized by Fe^{3+}, as is pyrite, but the rate is greater than that for pyrite for $[Fe^{3+}]$ greater than about 10 mg/L (Rimstidt et al., 1994).

Arsenic in MIW is often limited by the availability of arsenic (unlike sulfate) in the source material and by attenuation mechanisms. Scorodite is likely the most common secondary arsenic mineral and has low solubility under near-neutral pH conditions (Dove and Rimstidt, 1985). Sorption/co-precipitation of arsenic oxyanions onto positively-charged ferric-oxyhydroxide surface is well documented mechanism for removal of arsenic from acidic waters and may effectively limit arsenic concentrations in many MIW situations (Smith et al., 1992 and Gulens and Champ 1979). A tendency for arsenic to associate with ferric oxyhydroxides reinforces the importance of characterizing MIW with respect to turbidity as an indicator of the behavior of suspended ferric oxyhydroxides.

13.3.3 IRON AND ALUMINUM CONCENTRATIONS

Iron and aluminum associated with MIW often give rise to its most visible characteristic in the form of intense ochreous ferric oxyhydroxide "yellow-boy" staining of stream beds and conspicuous accumulations of white aluminum oxyhydroxides. The extremely low solubilities of these compounds under near-neutral conditions results in the persistence of such discoloration long after a MIW source ceases to be active.

Elevated iron concentrations are an obvious byproduct of the oxidation of pyrite or any other iron sulfide (see Figure 3). The salient aspects of aqueous iron chemistry are conveniently described by Eh-pH and solubility diagrams such as those presented by Hem (1985) which illustrate the instability of pyrite under oxidizing conditions at all pHs. Upon dissolution of pyrite, iron enters the aqueous environment as ferrous iron (Fe(II)) in the form of Fe^{2+} which is stable over a relatively wide range of Eh-pH conditions. As discussed in Section 13.3.1.1, in the presence of molecular oxygen and acidic conditions Fe^{2+} may be abiotically oxidized to ferric iron (Fe(III)) in the form of Fe^{3+}, or its hydrolysis products $FeOH^{2+}$ or $Fe(OH)_2^+$ (depending on pH). Fe^{3+} may also be photo-reduced to Fe^{2+} - a mechanism that may be especially important in high altitude, where ultraviolet radiation is more intense (Waite and Morel, 1984).

Rates of Fe^{2+} oxidation are dramatically enhanced by Thiobacillus ferrooxidans. Concentrations of dissolved iron, ferrous or ferric, are controlled by the precipitation of one or more ferric oxyhydroxide or oxyhydroxysulfate solid phases that have large stability fields in Eh-pH space. Under very acidic (e.g., less than pH = 1) and oxidizing conditions, significant concentrations (thousands of mg/L) of dissolved Fe(III) are possible in solutions in equilibrium with $Fe(OH)_3$, but under similar oxidizing conditions at pH 5, iron solubility is limited to 0.005 mg/L. Under less oxidizing conditions, where Fe^{2+} predominates over Fe^{3+}, the same solubility variations occur over only two pH units but very large concentrations of iron are possible at higher pHs (e.g., 1000+ mg/L at pH 4, Eh < 0.4 V). Smith et al. (1994) reports dissolved iron concentrations in a variety of 0.1 mm filtered-MIW samples to be a strong function of pH and to range between 0.0001 mg/L (pH 7-8) and 100,000 mg/L (pH <0).

Precipitation of Fe(III) solids from MIW is of considerable importance for aesthetic reasons, as a control on iron concentrations and as a potential control on other metal concentrations via sorption and co-precipitation. The mineralogy of these precipitates is quite complex and ranges from distinct phases such as goethite (aFeOOH) through less-well crystallized ferrihydrite ($Fe_5HO_8 \cdot 4H_2O$) and schwertmannite (also known as "mine drainage mineral" or MDM, $Fe_8O_8(OH)_6SO_4$) and others to essentially amorphous ferric oxyhydroxides (Murad et al., 1994). Seemingly a common attribute to all Fe(III) precipitates in MIW situations is their initially very fine particle size and tendency to form suspensions. Depending on aqueous environmental conditions these particles aggregate or flocculate and settle out of the water column or remain as colloid-sized particles and are preserved in suspension for long periods of time (Schmiermund and Ranville, 1996).

The mineralogy, fine particle size and associated large surface areas of freshly precipitated Fe(III) solids contribute to sorption/co-precipitation of heavy metals, resulting in metal enrichment of these solid phases (Bowell and Bruce, 1995). Transportation of suspended iron oxyhydroxides and associated heavy metals can represent a significant mode of metal mobility in MIW and one that differs in important ways from transport as dissolved metals. Flocculated material may result in the transfer of heavy metals to the stream bed and thus attenuate the metal flux from MIW sources but future floods may re-suspend the bed sediments causing increases in downstream metal fluxes in the absence of increases in MIW output. Colloidal material with sorbed MIW-derived heavy metals may transport those metals long distances (hundreds of kilometers) in concentrations exceeding that of the dissolved fraction (Kimball et al., 1995).

Aluminum is included with iron as a potential characteristic of MIW for a variety of reasons including its similar tendency to form oxyhydroxides and oxyhydroxysulfates in acid-neutralized environments and its potential to cause aesthetic degradation of MIW-affected waterways. Aluminum is also included with iron because, like iron, there is generally a readily available source for aluminum, albeit one that is very different from the iron sulfide minerals that give rise to iron in MIW. Aluminum in MIW is derived from acidic weathering of host rock and gangue minerals associates with many ore deposits and clay minerals in the enclosing sedimentary rocks of coal seams. Feldspars, micas and secondary clay minerals are generally unstable in strongly acidic environments and represent the primary sources of aluminum to MIW.

The aqueous chemistry of aluminum is extremely complex and prone to misunderstanding due in part to the tendency of aluminum to form dissolved polymeric species and colloidal macromolecules requiring special sampling and analytical handling to interpret correctly (Hem and Robertson (1990) and Sposito (1996). Smith et al. (1994) reports dissolved aluminum concentrations (in 0.1 mm filtered samples) to be a strong function of pH and to range between 0.01 (pH 7-8) and 10,000 mg/L (pH <0) in a variety of MIW. The control on aluminum solubilities in natural waters is generally thought to be amorphous ($Al(OH)_3$) although Nordstrom (1982) has proposed aluminum sulfate and hydroxysulfate as possible controls in acidic sulfate solutions. To the extent that amorphous $Al(OH)_3$ does control aluminum solubility, it would be expected that acidic MIW would form $Al(OH)_3$ precipitates upon neutralization and that dissolved aluminum concentrations would approach their minimum (< 0.001 mg/L) between pH 5 and 6. Kimball et al. (1994) and Crouse and Rose (1976) found this to be the case in a MIW-affected stream during artificial neutralization experiments and natural field conditions, respectively. In contrast, the solubility limit imposed by $Al(OH)_3$ at pH 4 is close to 100 mg/L. As in the case of ferric oxyhydroxides, aluminum oxyhydroxides may be present as suspended matter in the water column adding to turbidity.

13.3.4 HEAVY METAL CONCENTRATIONS

Elevated concentrations of one or more heavy metals are characteristic of MIW from virtually all metal mines and represent threats to downstream aquatic life forms as well as to humans through direct consumption and agricultural and aquacultural bioacumulation (an example is provided by King, 1995). (Heavy metals in this context is taken to be all non-ferrous transition metals and the main group metals except aluminum, including Ga, In, Tl, Sn, Pb, Bi, Po and the actinides Th and U.) The vast array of metallic mineral deposits, their inherent metal contents and mineralogy and environmental signatures are described by du Bray (1995). In general, any metal present in a mineral deposit is subject to incorporation in MIW under certain geochemical conditions.

The majority of primary metal-bearing phases commercially mined are sulfides and, as such, are inherently unstable under oxidizing conditions as is pyrite. Their oxidation mechanisms are also analogous to that of pyrite but reaction rates may be very different and are typically slower than that of pyrite as shown by Rimstidt et al. (1994). Most low-solubility secondary minerals of transition metals are carbonates or hydroxides and are unstable at low pHs due to their tendency to react with protons. Consequently, the highest aqueous concentrations of many metals in MIW are associated with oxidizing, acidic conditions as is evident from the data of Smith et al. (1994). However, this correlation does not imply that acidic or extremely acidic pHs, are necessary for a MIW to carry significant (with respect to regulatory limits and/or environmental health and safety limits) metal loads.

In ordinary (non-MIW) natural waters, heavy metals, as well as iron and aluminum, are typically thought of as trace metals as compared to the much more abundant alkali and alkali earth metals (Ca, Mg, Na and K). Many exceptions to this generality exist as can be seen from the examples provided by Hem (1985). However, in MIW, especially acidic ones, certain heavy metals may be present in major dissolved concentrations and in extreme cases (e.g. Iron Mountain, CA, see Alpers et al. (1992)) equaling or exceeding those of iron or aluminum and the alkalis and alkali earth metals.

Controls on heavy metal concentrations in MIW is a complex issue and is highly metal-specific and site-specific. Initially (at the point of MIW formation), the abundance of metal-bearing mineral phases as well as their solubilities and rates of dissolution control the concentrations of heavy metals. As waters evolve in response to increased contact with country rock, the atmosphere and other waters, changes in metal complexation may take place, precipitation may occur and sorption reactions are possible. In general, these reactions serve to attenuate metal fluxes. In many situations a clear understanding of the processes is required to prevent inadvertent worsening of an existing situation by future activities.

13.3.5 TURBIDITY AND SUSPENDED MATTER

Turbidity is the ability of a water to disperse and absorb light so as to prevent straight-line transmission and is caused by suspended particles greater than about one nanometer in diameter. Turbidity is a in-field measurable parameter and is directly related to visual appearance of a water. Total suspended solids is a laboratory-measured parameter and has the distinct disadvantage, relative to turbidity, of involving sample storage. During storage, reactions, such as precipitation and flocculation, may occur and irreversibly change the sample relative to field conditions. Particles relevant to MIW include macromolecular through colloid-sized particles of iron and aluminum oxyhydroxides, macroscopic particles of flocculated metal oxyhydroxides and other compounds maintained in suspension by turbulence and normal suspended clays and silts due to erosion.

Turbidity and total suspended solids are important characteristics of MIW because they may have associated regulatory MPLs and because of their relevance to transport of heavy metals and arsenic and their obvious impact on iron and aluminum concentrations. The importance of collecting both unfiltered and filtered water samples cannot be overemphasized in this connection. Furthermore, distinguishing between total and dissolved iron and aluminum requires filtration through at least a 0.1 mm filter. The 0.45 mm filter media typically used for this purpose is inadequate and has absolutely no relevance to aqueous chemistry.

Depending on local regulatory requirements, either or both total and/or dissolved metal concentrations may be relevant. Turbidity may define where certain metal-attenuation reactions are occurring in a MIW watercourse and provide important clues for remediation planning and criteria for water use. Careful attention should be given to prevailing definitions of maximum permissible limits (MPLs) that may or may not address filtered versus unfiltered concentrations or prescribe filtration procedures. Regardless of regulatory requirements, an accurate characterization that is necessary for a functional understanding of MIW chemistry requires that the distinction be made.

13.4 MIW REMEDIATION COSTS

by M. A. Drozd

13.4.1 BASIC ESTIMATION ASSUMPTIONS

For the purpose of estimating the cost of treating MIW under various treatment schemes, all economic analyses will use the same base assumptions. The material to be analyzed will be 2.5 million tons of waste with 0.5% pyrite present along fracture planes exposed by blasting. Approximately 50% of the pyrite will be on exposed surfaces. It is assumed that the reactive portion of the material will take 50 years to be depleted and no further acid potential above natural geologic levels will be present after that time. The rainfall for the area is 8 inches per year and approximately all of this material will be spread evenly over each year's period. This assumption is valid if you consider that most pits will produce ground water. The area of surface run-off is 100 acres. The water quality of the effluent is:

Parameter	Concentration
pH	3.2
Fe^{+2}	10.0 g/l
Fe^{+3}	1.0 g/l
Al^{+3}	15.0 ppm
Cu^{+2}	1.0 ppm

No arsenic or selenium are present in the ore. Treatment of the water exiting from the property prior to cessation of mining is treated and discharged by ordinary chemical treatment methods except in the case of bio-remediation. Solution impoundment is minimal so that surface evaporation is not considered. All sludge produced is solid waste requiring disposal at a licensed land fill at a cost of $0.75 cubic foot (about $50 per ton at density of 65 lbs/ft^3). The transportation cost is included in the disposal cost. Project life is seven years.

13.4.2 CHEMICAL TREATMENT

13.4.2.1 Chemical Treatment Estimate

The water flow from the project is:

100 acres x 0.75 feet per year = 75 acre-feet of water

75 acre-ft x 325,851 gal/acre-ft = 24,438,825 gal/yr

24,438,825 gpy / 365 d/y / 1440 min/day = 46 gpm

The treatment process used is a two stage hydroxide precipitation followed by pH modification to 8.5 for release. Lime is used for pH modification in the form of lime water and milk of lime. Sludge density technology, such as HDS, is used along with drying of the sludge. Extrusion drying is utilized to further compact the material. Aluminum removal is accomplished at a pH level of 7.8., while copper and iron removal requires a pH level of 9.0. Clarification of the material is accomplished by the HDS system and a small sampling pond (115' x 115' @ depth of 11' with 3 to 1 side slopes) containing approximately 5 days retention with 2' of freeboard. The project will operate for 7 years with 43 years of additional treatment.

13.4.2.2 Capital Cost Estimate

Plant Feed Holding Pond - 75' x 75' x 8' (including excavation, double lining, 60 mil HDPE, monitoring system and engineering)	$15,000
Feed pump (2 pumps)	$10,000
Plant Holding Tank (10'∅ x 10')	$15,000
Treatment Tank 1 (5'∅ x 5') (Rubber Lined)	$5,000
Agitator (Stainless Steel)	$1,000
Treatment Tank 2 (5'∅ x 5')	$4,000
Agitator (Stainless Steel)	$1,000
pH Adjustment Tank (5'∅ x 5') (Rubber Lined)	$5,000
Agitator (Stainless Steel)	$1,000
Building (50' x 25' x 15' eave height)	$63,000
Lime Water Tank (10'∅ x 10')	$15,000
Lime Water Pump	$1,500
Agitator	$5,000
Milk Of Lime Tank (6'∅ x 6')	$6,000
Agitator	$1,000
Milk of Lime Pump	$2,500
Sulfuric Acid Pump	$2,000
Sump Pump	$8,000
Dryer	$15,000
Clarifier (High Capacity 4' Diameter)	$20,000
Lime Bin and Feeder	$50,000
Piping	$10,000
Speciality Concrete Work and Concrete Coating	$17,000
Electrical	$20,000
Instrumentation	$20,000
Structural Steel	$20,000
Lab Furniture and Equipment	$50,000
Internal Carpentry	$12,000
Erection Cost	$35,000
Miscellaneous Equipment	$15,000
Finished Solution Holding Pond (110' x 110'x 11')	$25,000
Engineering	$23,000
Subtotal	$493,000
Contingency (25%)	$123,000
Total Capital Cost	$616,000

13.4.2.3 Chemical Treatment Operating Costs

13.4.2.2.1 Labor (Yearly)

2 men per shift @ $12.00/hr & 40% burden and 25% support $21.00/hr x 2 men x 24 hr/d x 365 d/yr $367,900

13.4.2.2.2 Power (Yearly)

Power Requirements 50 KVA
Power Cost (including Demand) $0.06/KWH
50 KVA x 24 hr/d x 365 d/y x $0.06/KWH
$26,300

13.4.2.2.3 Chemicals, Maintenance and Consumables (Yearly)

Lime Consumption - 10,625 lbs/day	
Lime Cost/lb - $0.05/lb	
Lime addition cost	$193,900
Sulfuric Acid	$500
Flocculent	$2,000
Maintenance and Supplies	$36,500
Propane	$5,000
Other Supplies	$9,000
Sludge Disposal Costs - $50/ton	
Sludge production - 5,124 tons per year	
Disposal cost	$256,200
Yearly Cost	$503,000
Contingency (25%)	$126,000
Total	$626,000

13.4.2.2.4 Total Yearly Chemical AMD Treatment Operating Costs

Labor	$368,000
Power	$26,000
Chemicals, etc.	$626,000
Total	$1,020,000

13.4.2.2.5 Surface Reclamation Costs

Topsoil placement and re-vegetation efforts will cost $2,400 per acre for side slopes and $1,800 per acre for top surfaces of heaps and dumps. Total tons of ore mined are 3.5 million tons. Ore tonnage is 1,000,000 tons. The mining method is open pit mining. The waste to ore ratio is 2.5 to 1. The pit occupies 12 acres. The waste dump, at maximum capacity, occupies 25 acres. The pad and plant area occupies 43 acres. An additional 20 acres is disturbed for roads, ore storage, laydown areas and offices. The total site disturbance to be reclaimed is 100 acres. The sludge generated during mine water treatment does not pass a TCLP (toxic leach characterization procedure) test and must be disposed in a licensed landfill. The waste dump material density is 120 lbs/ft^3 and ore density (on the heap) is 100 lbs/ft^3. The total disturbed mine surface area subject to reclamation is 100 acres. The road, ore areas, shops, offices and borrow areas will be reclaimed with the pads. The annual precipitation for the area is 8 inches per year.

Top Surface Contouring and Topsoil Placement Cost	$2,500/acre
Top Re-vegetation Costs	$1,800/acre
Total Cost/acre	$4,300/acre

Cost Estimate:

Re-vegetation 100 acres x $4,300/acre	$430,000
Surface grading	$420,000
Heap Detoxification	$400,000
Total	$1,250,000

Total Reclamation Cost:

Water Treatment Plant Capital	$616,000
Water Treatment Operating Cost for 50 yrs	$51,000,000
Surface Reclamation	$1,250,000
Total Reclamation Cost	$52,866,000

(All closure cost estimates are shown with as much cost detail as possible to allow manipulation of the values for site specific parameters.)

13.4.2.3 Semi-passive MIW Treatment Cost Analysis

13.4.2.3.1 Semi-Passive Treatment Scheme

The semi-passive treatment scheme includes reducing "bog" inoculated with sulfate reducing bacteria (anaerobic pre-treatment), Anoxic Limestone Drain (pH modification), SRB treatment structure (metal removal), and constructed wetlands (metal, BOD and toxin polishing). All pond structures are underlain with a double PVC liner and with inter-liner leak detection. The PVC liner is covered with 2' of dirt. Internal treatment requirements, such as hay and manure fill, etc., are built on top of the dirt fill. Each structure is built with a smaller by-pass facility to allow for cleaning and maintenance. Maintenance of the biological structures is preformed by an outside contractor. The contractor is responsible for maintaining all bacterial cultures and doing all bacterial culturing tests. The service supplies two bacterial re-augmentations per year for each system. Additional re- augmentations are billed at $7,500 each. The cost of this service is $90,000 per year. All material removed from the structures must be disposed in a solid waste facility (cost $50/ton). ALD's are renewed every three years, SRB are renewed every 10 years, and the constructed wetlands is renewed every 12.5 years. Renewal involves removal and disposal of spent material and complete replacement of consumable material. All structures are excavated to allow pooling and overflow movement of the treated solution. Solution flow design allows up to 250% excess to flow into the system except in the by-pass mode. Any flow in excess of the 250% (or 46 gpm when in the by-pass mode) will be diverted into an overflow pond and pumped back to the system when possible. The overflow pond will have a single liner and a capacity of 25% of the annual precipitation (approximately 6 million gallons).

13.4.2.3.2 Pre-Treatment SRB Pond Cost Estimate

Pre-treatment SRB Pond - 75' x 75' x 8' (including excavation, double lining, 40 mil PVC, monitoring system and engineering)	$25,000
3500 cubic feet of peat, manure and hay installed	$10,000
Cement work and gravel packing for inlet and outlet structures and drain piping	$25,000
Initial culture isolation, pilot work and initial inoculation	$40,000
Construction management, design engineering quality assurance	$20,000
Subtotal	$120,000
Contingency (25%)	$30,000
Total	$150,000

13.4.2.3.3 Anoxic Limestone Drain Cost Estimate

Pond structure for 115,000 cubic feet of limestone, 100' x 175' 12 deep. Structure includes a geotextile, 60 mil HDPE top and coated pre-stressed concrete baffle walls sealed with bento-mat. A recording flow device is placed on the in-flow and exit to monitor flow characteristics of system. The structure is entirely excavated, and the pond's side slopes are 3 to 1. $192,500

11,500 tons of limestone @ $45/ton	$517,500
Ancillary by-pass ALD (all costs included)	$50,000
Flow metering system and remote alarms	$15,000
Subtotal	$775,000
Contingency	$195,000
Total	$970,000

13.4.2.3.4 SRB Treatment Cost Estimate

SRB treatment structure consists of a pond with internal baffling that can how the solution with 250% excess flow (flood conditions). Design utilizes upflow design.

Pond - 250 x 250 x 12' including double 40 mil PVC liner, leak detection system. Internal free standing pre-stressed concrete baffles.	$195,000
Organic substrate @ $1.75 per ft^3	$775,000
Engineering, Procurement and Construction Management	$75,000
By-pass system all costs included	$250,000
Quality Assurance	$15,000
Subtotal	$1,310,000
Contingency	$328,000
Total	$1,638,000

13.4.2.3.5 Constructed Wetlands Cost Estimate

Six acres (24,300 m^2) of constructed wetlands will be built to polish the water exiting the SRB/ALD/SRB system. The TVA estimated a constructed wetland cost from $3.58 to $32.03 per square meter, in 1986. A mid-point cost of $17.80 and a ten year inflation factor of 33% (2.9% per year) would give a 1996 estimated cost of $23.67 per square meter.

24,300 m$_2$ @ $23.67	$575,000
By-pass System	$125,000
Total	$700,000

13.4.2.3.6 Annual and Recurring Cost Estimate

Contract maintenance	$90,000
per year for 50 years	$4,500,000
ALD renewal (every 3 years)	
Limestone ($517,500 x 17)	$8,789,000
Limestone disposal (11,500 tons x $50 x 17)	$9,775,000
SRB renewal (every 10 years)	
Substrate ($775,000 x 5)	$3,875,000
Substrate disposal ($0.75/ft^3) x 5	$1,650,000
Renewal of constructed wetlands (every 12.5 years)	
Renewal cost ($575,000 x 4)	$2,300,000
Spent wetlands material disposal (5,000 tons) 5000 x $50/ton x 4)	$1,000,000
Total maintenance and recurring costs	$31,889,000

13.4.2.3.7 Surface Reclamation Costs

Topsoil placement and re-vegetation will cost $2,400 per acre for side slopes and $1,800 per acre for top surfaces of heaps and dumps. The total ore/waste mined tonnage is 3.5 million tons. Ore tonnage is 1,000,000 tons. The mining method is open pit mining. The waste to ore ratio is 2.5 to 1, and the pit occupies 12 acres. The waste dump at maximum capacity occupies 25 acres. The pad and plant area account for 43 acres. An additional 20 acres is required for roads, ore storage, laydown areas and offices. The total site area subject to eventual reclamation is 100 acres. The road, ore areas, shops, offices and borrow areas will be reclaimed with the pads. The sludge generated during mine water treatment does not pass a TCLP (toxic leach characterization procedure) test and must be disposed in a licensed landfill. The waste dump material density is 120 lbs/ft^3 and ore density (on the heap) is 100 lbs/ft^3. An additional area of 20 acres is associated with biological treatment.

Top Surface Contouring and Topsoil Placement Cost	$2,500/acre
Top Re-vegetation Costs	$1,800/acre
Total Cost/acre	$4,300/acre
Cost Estimate:	
Re-vegetation 120 acres x $4,300/acre	$516,000
Surface grading	$420,000
Heap Detoxification	$400,000
Total	$1,336,000

13.4.2.3.8 Estimated Total Semi-passive Treatment Cost

Capital Costs	
Pretreatment SRB	$150,000
ALD	$970,000
SRB Treatment	$1,638,000
Constructed Wetlands	$700,000
Operating Costs	$31,889,000
Surface Reclamation	$1,336,000
Total	$36,683,000

13.4.2.4 MIW Material That Has "Gone Acid"

13.4.2.4.1 Isolation Scheme

The MIW material location in the waste dump is not exactly known. Consequently, all material will have to be moved to a new location. No infrastructure is present on site since the MIW production was discovered during post closure monitoring. All work is done by a contractor. The MIW material is identified using wipe tests on the material and sight observation by a geologist. Any questionable material will be moved to the isolation area. Material moving costs are $4.00 per yard. Contract geologists are retained at $70,000 per year and are retained by the original operators to provide quality assurance. A contract reclamation superintendent is hired at $100,000 per year. Truck, office and sundry costs are $200/day. Work proceeds 250 days per year and only eight hours per day (during daylight).

Closure design utilizes the open pit that was not back filled. The pit contains water derived from ground water flowage and precipitation. The ground water movement has been hydrologically shown to be relatively slow, and would safely allow sub-aqueous disposal of the material. A drainage system is designed for the pit bottom to allow additional pit water pump down, if design parameters prove incorrect. Heap material has started to "go acid". Isolation design allows for capping of the material in-place. All previous re-vegetation work must be removed and redone.

The ore is only 28.6% of the material mined. All the waste material cannot be back-filled into the pit. Non-acid producing material will be moved to a new surface disposal area to be covered and re-vegetated. Each cubic yard of waste weighs 1.6 tons. The time period for the new closure procedure is two years. During the reclamation work a portable water treatment plant will treat all on site precipitation. Water treatment plant rental is $25,000 per month, and holding pond construction costs are $65,000. Water treatment capital costs are the same as the Chemical Treatment estimate. Engineering and design for an approved treatment plan requires one year and costs $500,000. Upon state approval of permits, the waste dump is neutralized and the bacteria are killed using a bactericide. The bactericide usage is maintained throughout the closure work. The treatment plant is returned when the bacteria are killed. The ponds are used for the neutralization of the waste dump. Capping of the heaps commences as soon as possible upon receiving approval of the remediation plan. The reclamation work done during closure will cost $1,250,000.

13.4.2.4.2 Heap Reclamation Costs

Heap closure includes adding a bactericide to the heap prior to capping. Capping is done by removing all topsoil from the heaps. The side slopes of the heap are graded 5 to 1 where possible. Excess material is removed for screening and pH modification. The re-graded surface is then compacted and smooth graded. The heap is covered with a rough surface HDPE liner underlain by a 10 oz. geotextile. The liner is then covered by fines (-1/2") screened from the heap material. The +1/2" material is then placed on top of the fine material as a root barrier. The top soil is replaced on the root barrier.

Bactericide Treatment $700 per acre	
Heap Area - 8 acres x $70	$6,000
Remove topsoil	$25,000
Heap material removal 100,000 yards	$400,000
Grading	$66,000
Screening	$50,000
HDPE liner	$292,000
Geotextile	$32,000
Replacing screened material($3/yd^3)	$300,000
Replace topsoil and modify pH	$35,000
Re-vegetation ($1,800 per acre) x 8	$15,000
Subtotal	$1,221,000
Contingency (25%)	$305,000
Total	$1,526,000

13.4.2.4.3 Mine Waste Remediation

Water treatment - for one year	
Mobile plant rental $25,000/month	$300,000
Water treatment labor and power	$460,000
Water treatment chemicals	$503,000
Pond construction	$65,000
Topsoil removal	$162,000
Bactericide treatment and stabilization	
($1,200 per acre) x 25 acres	$30,000
Caustic neutralization of waste dump	$576,000
Geologic Quality Assurance	$280,000
Site Supervision	$100,000
1,562,500 cu yd moved	$6,250,000
Compaction	$260,000
Capping of Pit Liner 40 mil PVC	$1,020,000
Root barrier	$1,800,000
Fine cover	$484,000
Topsoil replacement/pH modification	$200,000
Re-vegetation ($1,800/acre) x 25	$45,000
Non-acid producing waste dump - area 8 acres	
Surface preparation	$350,000
Re-vegetation ($4,300/acre includes grading)	$35,000

Pit underdrain	$100,000
EPCM	$500,000
Subtotal	$13,520,000
Contingency (25%)	$3,370,000
Total	$16,890,000

13.4.2.4.4 Total Reclamation Cost Estimate for Post-closure Remediation

Chemical costs	$1,020,000
Heap Closure	$1,526,000
Previous Reclamation	$1,250,000
Mine Closure	$16,890,000
Total	$20,686,000

13.4.2.5 MIW Material That Has Not "Gone Acid"

13.4.2.5.1 Remediation Scheme

The exact location and quantity of MIW material on site is not exactly known. Therefore, all material will have to be moved to a new location. Mining equipment and infrastructure is still in place because the mine has just stopped operations. The MIW potential has been defined in the ore and the operator wishes to address the reclamation potential at this juncture. All work is done by mine personnel using equipment already at the site. MIW material is identified using wipe tests on the material and sight observation by a geologist. Any questionable material will be moved to the isolation area. Material moving costs are $1.50 per cubic yard. An ore control geologist is kept at a cost of $49,000 per year. A reclamation superintendent, the previous general manager, is also retained at a salary of $105,000 per year. Sundry costs are $200 per day. Work proceeds 250 days per year and only eight hours per day (during daylight).

Closure design utilizes the open pit for a back fill location. The first step is to deal with the water in the pit. A drainage system is designed for the pit bottom to allow additional pit water pump down, if design parameters prove incorrect. Heap material has started to "go acid". Isolation design allows for capping of the material in-place. All previous re-vegetation work must be removed and redone.

All waste material cannot be back-filled into the pit. Non-acid producing material will be moved to a new surface disposal area to be covered and re-vegetated. The time period for the new closure procedure is two years. During the reclamation work no water treatment will be necessary. Engineering and design for an approved treatment plan requires six months and costs $200,000.

13.4.2.5.2 Heap Reclamation Costs

Heap closure includes capping. Capping is done by removing all topsoil from the heaps. The side slopes of the heaps are graded to 5 to 1 where possible. Excess material is removed for screening and pH modification. The re-graded surface is then compacted and smooth graded. The heaps are covered with a rough surface HDPE liner underlain by a 10 oz. geotextile. The liner is then covered by the fines (-1/2") screened from the heap material. The +1/2" material is then placed on top of the fine material as a root barrier. The top soil is replaced on the root barrier.

Remove topsoil	$25,000
Heap material removal 100,000 yards	$150,000
Grading	$30,000
Screening	$30,000
HDPE liner	$292,000
Geotextile	$32,000
Replacing screened material ($0.75/yd^3)	$75,000
Re-vegetation ($1,800 per acre) x 63	$113,000
Subtotal	$747,000
Contingency (25%)	$187,000
Total	$934,000

13.4.2.5.3 Mine Waste Remediation

Geologic Quality Assurance	$198,000
Site Supervision	$105,000
1,562,500 cu yd moved	$2,344,000
Compaction	$100,000
Capping of Pit:	
Liner 40 mil PVC	$1,020,000
Root barrier	$1,800,000
Fine cover	$484,000
Topsoil replacement/pH modification	$200,000
Re-vegetation ($1,800/acre) x 25	$45,000
Non-acid producing waste dump - area 8 acres:	
Surface preparation	$350,000
Re-vegetation ($4,300/acre includes grading)	$35,000
EPCM	$200,000
Subtotal	$6,881,000
Contingency (25%)	$1,720,000
Total	$8,601,000

13.4.2.5.4 Total Reclamation Cost Estimate for Post-closure Remediation

Heap Closure	$934,000
Mine Closure	$8,601,000
Total	$9,535,000

13.4.2.6 MIW Control During Mining

Sulfur analysis of all samples taken during exploration, development, and operation, plus data collected from the computer mine models, coupled with mine face geologic screening can provide the information needed by the mine operators to segregate all potential MIW waste. The potential MIW waste will then be dispatched to a special handling structure built within the waste dump. The structure would be located in clay lenses that have a minimum of three feet of clay on all sides of the in-place MIW material. A sulfur ore control geologist would have to be retained at a cost of $49,000 per year during the mine life. The sulfur analysis cost would be approximately $1.50 per sample. Over the life of the project the extra sulfur analysis would be $1,250,000. Clay occlusion of the waste would cost $500,000. At the end of the project, closure would require cyanide detoxification, grading and re-vegetation. The estimated cost is $1,250,000. The total reclamation cost would be:

Geological manpower	$343,000
Sulfur analysis	$1,250,000
Clay isolation	$500,000
Surface Reclamation	$1,250,000
Total	$3,343,000

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