

Sludge management risks can include low sludge density, lack of appropriate on site disposal, off-site transportation and disposal issues, poor sludge stability (chemical mobilization, physical instability), sludge pond access risks (human/fauna), and dusting (airborne contamination). Risks of final site conditions must also be assessed, such as the risk of natural disasters to the treatment system (e.g., earthquake, excessive precipitation).

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7.2 Objectives of Mine Drainage Treatment

The objectives of mine drainage treatment are varied and may include one or more of the following:

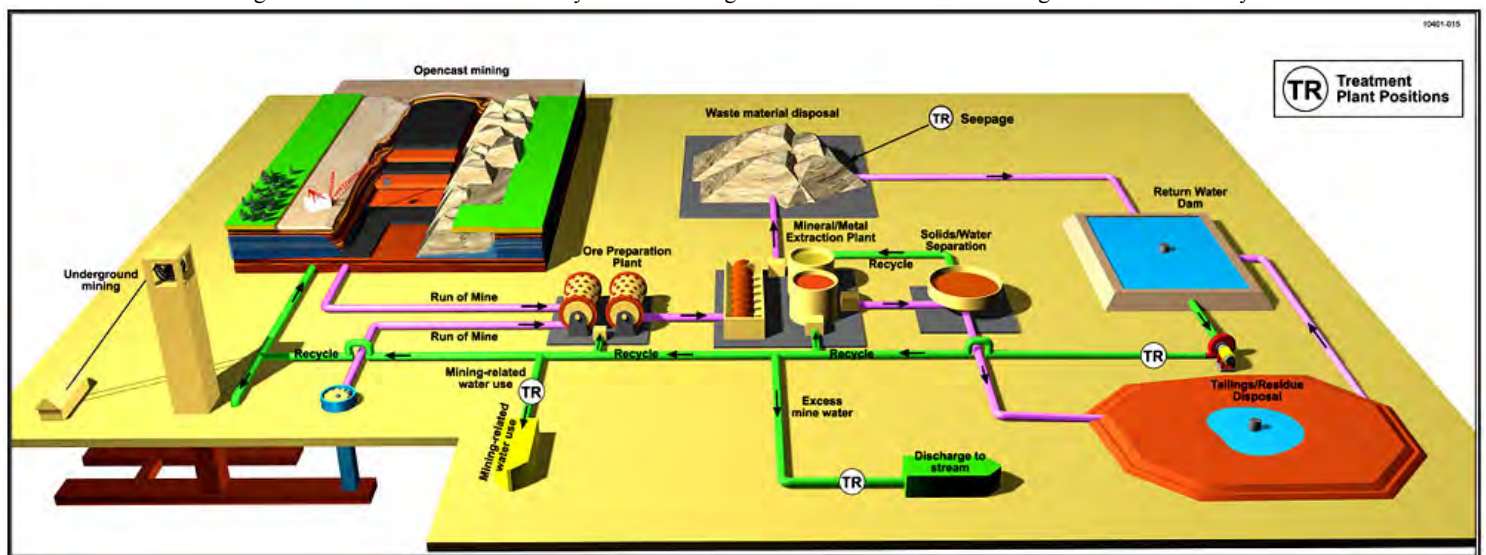
- Recovery and reuse of mine water within the mining operations for processing of ores and minerals, conveyance of materials, and operational use (e.g., dust suppression, mine cooling, and irrigation of rehabilitated land). Most mining operations include the management of water on the mine site and manage associated water infrastructure. The mine water balance requires management of different demands and sources for water volume and water quality. Mine drainage treatment, in this case, is aimed at modifying the water quality so that the treated effluent is fit for the intended use on the mine complex or site. Where multiple water sources are available it is typically less costly to keep the water sources separate to reduce the volume of water to be treated. This option is particularly true when off-site run-off water can be diverted away from the mine and waste facilities to reduce water volume needed to be treated.
- Protection of human health in situations where people may come in contact with the impacted mine water through indirect or direct use of mine water drainage.
- Environmental protection, specifically related to mining water impacts on surface water and groundwater resources. Mine drainage may act as the transport medium for a range of pollutants, which may impact on-site and off-site water resources. Water treatment would remove the pollutants contained in mine drainage to prevent or mitigate environmental impacts.
- Useful and potentially saleable products may be recovered from mine drainage. It is unlikely that by-products recovery would be a sole driver to the installation of a water treatment facility. However, when commodity prices are high, the recovery of saleable products will improve the financial viability of mine drainage treatment projects.
- Regulatory requirements may stipulate a mine water discharge quality or associated discharge pollutant loads. Any discharge of mine drainage to a public stream or aquifer must be approved by the relevant regulatory authorities. Discharge quality standards may not be set for many developing mining countries, but internationally acceptable environmental quality standards may still apply as stipulated by project financiers and company corporate policies.
- Mine water is a valuable resource and much of the world is facing water stress. The beneficial use of mine water to satisfy the needs of a variety of mining and non-mining water users can be a key driver supporting the installation of mine drainage treatment facilities. There is an increasing number of mine drainage treatment projects aimed at supplying treated mine water to neighbouring communities and industries around mines.
- Sustainability of mining will require the mitigation, management, and control of mining impacts on the environment. In many cases, the mining impacts on water resources are long term and persist in the post-closure situation. Mine drainage treatment may be a component of overall mine water management to support a mining operation over the mine's entire life and enhances post-closure and sustainable use of the mine property long after the ore deposit is depleted.

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7.3 Mine Drainage Treatment

The approach to mine drainage treatment is based on an understanding of the integrated mine water system and circuits and the specific objective (or objectives) to be achieved. A generic mine water system diagram is shown in Figure 7-1 to demonstrate the point that treatment may be introduced at several different points or locations on a mining project and to illustrate different purposes and objectives.

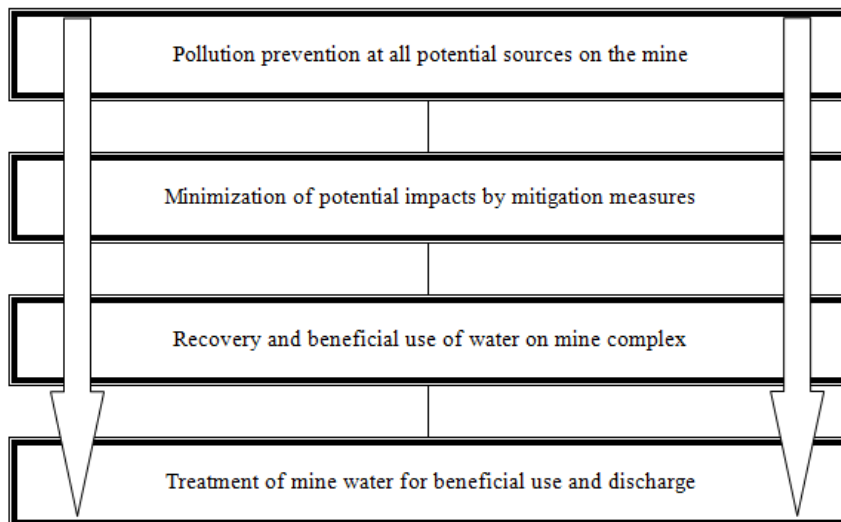
Figure 7-1: Generic Mine Water System Indicating Potential Position for a Drainage Treatment Facility



The generic location for a mine drainage treatment facility includes the following:

- A selected mine water stream originating from a process or facility discharging high concentrations and loads of pollutants
- A water stream dedicated to some mining-related water use, which may require a specific water quality
- A return water stream to render the recycled water fit for use in the mining or minerals processing operation
- A point or diffuse discharge stream to a natural watercourse or aquifer

Mine drainage treatment projects are executed within the overall hierarchy of mine water management, which generically includes the following steps:



This approach adopted for mine drainage treatment will be influenced by a number of considerations related to the following:

- Before selecting the treatment process, a clear statement and understanding of the objectives of treatment should be prepared. Mine drainage treatment must always be evaluated and implemented within the context of the integrated mine water system. Treatment will affect the flow and quality profile in the water system; therefore, the sized treatment system is selected based on mine water flow, water quality, cost, and ultimately water uses.
- Characterization of the mine drainage in terms of flow and key properties of ARD, NMD, or SD should include careful consideration of temporal and seasonal changes. Flow data are especially important because this information is required to properly size any treatment system. Particular concern should be taken to account for extreme precipitation and snowmelt events to ensure that the collection ponds and related piping and ditches are adequately sized and maintained. The key properties of mine drainage relate to acidity and alkalinity, sulphate content, salinity, metal content, microbiological quality, and the presence of specific compounds associated with specific mining operations, such as cyanide, ammonia, nitrate, arsenic, selenium, molybdenum, and radionuclides. Coal mine drainage (CMD) typically contains iron, aluminum, and manganese in significant concentrations. Other metals are usually only present in trace concentrations, and as mentioned in Chapter 2, these are usually removed in the process of meeting the typical CMD standards for manganese. There are also a number of properties of the mine-drainage constituents (e.g., hardness, sulphate, and silica) that may not be of regulatory or environmental concern in all jurisdictions currently, but that could affect the selection of the preferred water treatment technology.
- Different stages of mining and how the mine water system and water balance will change over the life of a mine. A mine drainage treatment facility must have the flexibility to deal with increasing and decreasing water flows, changing water qualities, and regulatory requirements. This may dictate phased implementation and modular design and construction of a treatment facility. Additionally, the post-closure phase may place specific constraints on the continued operation and maintenance of a treatment facility.
- Commodity-specific water aspects related to compounds present in the mine drainage (e.g., presence of radionuclides in the case of uranium mining). Some mining or processing operations may introduce extraneous chemicals and reagents into the water circuits. Reagents from one minerals processing plant (e.g., copper recovery) may be detrimental to another minerals processing plant (e.g., phosphate recovery).
- Practical mine site features, which will influence the construction, operation, and maintenance of a mine drainage treatment facility, including the following:
 - Mine layout and topography
 - Space
 - Climate
 - Sources of mine drainage feeding the treatment facility
 - Location of treated water users
- Handling and disposal of treatment plant waste and residues, such as sludges and brines

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7.4 Drainage Sources, Collection and Management

There are several main types of drainage that may require treatment before discharge from a site: acidic drainage, neutral drainage, and saline drainage. Each type of drainage, while distinct in its typical composition and chemistry, can typically be treated using similar, if not, identical treatment technologies. Chapter 2 provides more detail on the compositional characteristics of these mine waters. Certain mine waters, for instance from coal operations, may contain specific constituents that are challenging to treat, such as selenium. When certain constituents are absent, for instance iron in neutral drainage, chemical treatment of other parameters is often more difficult.

Drainage sources include waste rock dumps, tailings impoundments, haulage roadways, milling areas, contaminated surface, and underground mine workings. One of the most critical steps in any site treatment strategy is the water management plan. A critical component of treatment systems design is the flow rate. By decreasing annual flows requiring treatment, this will decrease operating and capital costs for the system. The key to an effective water management plan is to divert clean water and concentrate contaminated waters requiring treatment.

The objectives of a water management system are (Aubé and Zinck, 2009):

- To ensure diversion of all attainable uncontaminated waters using ditches and berms on upper water catchment areas
- To ensure capture of all contaminated waters
 - If contaminated waters come in contact with clean water, the clean water becomes dirty and volumes of water to be treated increase
 - Prevent release of contaminated water
- To minimise footprint and contact
 - Smaller waste storage and processing areas will minimise contact and result in more clean water
 - Covered waste piles prevent contamination

The water management system components and infrastructure pose engineering and operational challenges because of the variable flow rates and the corrosive or scaling nature of mine drainage. The considerations in the development of a mine drainage collection and conveyance system include the following:

- Properties of mine drainage, including corrosiveness, scale/precipitate forming potential, solids deposition, organic fouling, and plugging
- Dealing with variable mine drainage flows and qualities as dictated by climatic and seasonal changes and by the different stages of the life of the mine (The sizing of collection ponds and ditches is particularly critical where combined snow or precipitation events can combine to over top and cause failure of these facilities)
- The size of the collection ponds and ditches may be defined by the regulatory requirements (i.e., to meet a 24-hour 100-year precipitation event)
- Site and route selection based on consideration of topography, geotechnical conditions, and climate
- Selection of appropriate materials of construction
- Engineering features, including pretreatment before conveyance, pumping installation, and piping systems
- Operational aspects related to access, regular cleaning, monitoring, typical failures, and risks
- Maintenance aspects, particularly ease of cleaning

Mine drainage diversion, collection and conveyance systems are critical components of any treatment project. Appropriate basis of design must be developed and integrated into the overall treatment project. Surge ponds may be a valuable feature in the case of highly variable mine drainage flows and pollutant loads. This will afford some protection against surcharging the treatment system. It is typically not economical to build very large raw water retention ponds nor is it economical to build small ponds and very large treatment plants. Optimum sizing of both must be done together to determine best cost/efficiency ratio. Examples exist of failed projects because of the neglect of the design, operation and maintenance of the mine drainage collection infrastructure.

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7.5 Mine Drainage Treatment Technologies

A wide spectrum of drainage treatment technologies has been developed, proven, and applied to many different applications. The generic range of mine drainage treatment technologies is reflected in Figure 7-2. The description of the different drainage treatment technologies in this section will be framed in the context of current best practice of proven technologies.

Mine drainage treatment technologies can be broadly classified into active treatment, passive treatment, and in situ treatment as described in Table 7-1. The selection of the appropriate category of mine drainage for a specific application is influenced by the aspects summarized in Table 7-1

Figure 7-2: Generic Range of Drainage Treatment Technologies

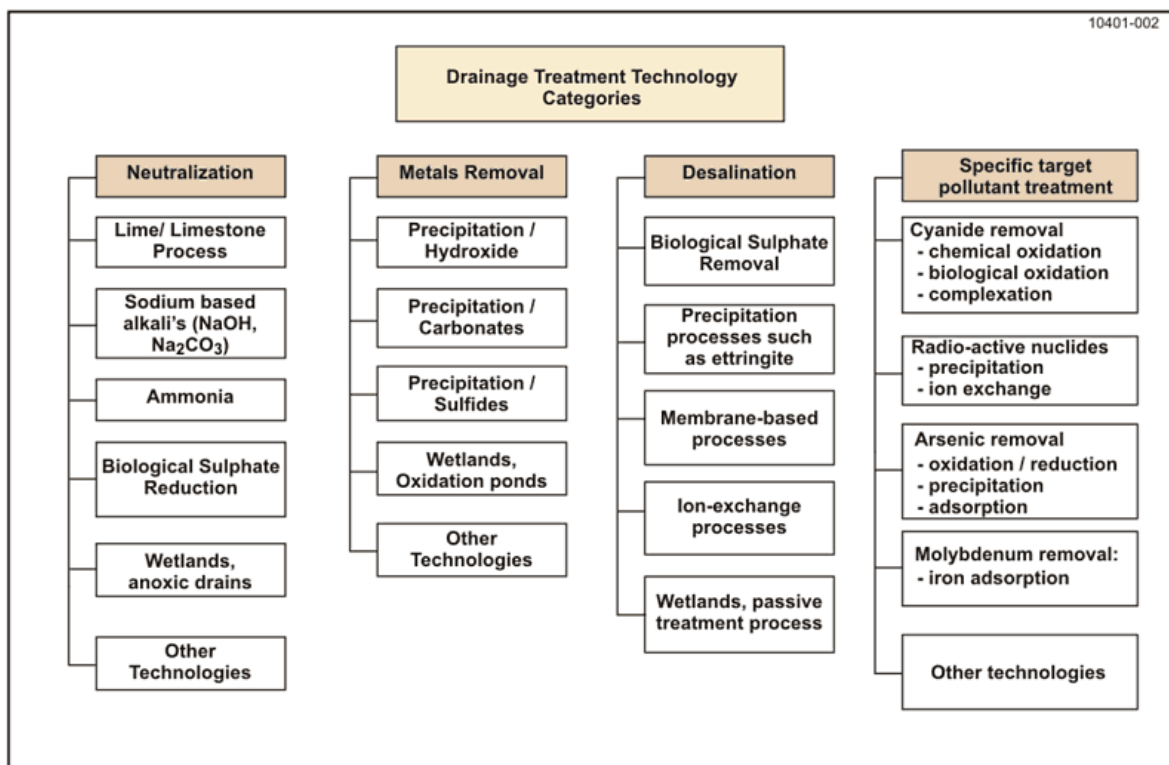


Table 7-1: Qualitative Comparison of Different Categories of Treatment

Feature / Characteristic	Active Treatment	Passive Treatment	In Situ Treatment
1. Application to phase of mining	Most appropriate to exploration and operational phases because it requires active control and management. Closure and post-closure applications mainly associated with large flows.	Most attractive to the closure and post-closure phases, because it requires only intermittent supervision, maintenance, and monitoring of self-sustaining processes.	Appropriate to the exploration and operational phases because it requires ongoing operation and maintenance.
2. Operational involvement	Active and ongoing plant operations and maintenance systems and personnel.	Constant operations not required, but regular maintenance essential.	Active and ongoing operational personnel required, but permanent presence on site not required.
3. Operational inputs and materials	Requires chemicals, operations staff, maintenance staff, electrical power, continuous and/or regular monitoring.	Self sustaining processes, periodic maintenance, intermittent monitoring. May require replacement or supplement of materials at low frequency.	Requires chemicals, operations staff, intermittent field maintenance, electrical power and low frequency monitoring.
4. Supply of power	Electrical and mechanical energy sources.	Natural energy sources of gravity flow, solar energy and bio-chemical energy.	Electrical and mechanical energy sources.
5. Management and supervision requirements.	Ongoing management engagement, constant facility supervision.	Low level management engagement and low frequency intermittent supervision.	High frequency supervision, but no permanent site presence required.
6. Range of application: <ul style="list-style-type: none"> ▪ Flow rates ▪ Constituents of interest 	Application to all flow rates, especially high flow rates and any constituent of interest.	Mainly applied to low flow rates and acidity, metals, and sulphate removal.	Large spectrum of volume and flow applications, mainly to deal with acidity and metals removal.
7. Treated water quality	Treatment process can be purpose built to deal with spectrum of treated water requirements.	Treated water quality poorer and more variable than other options.	Treated water quality lower and more variable than active treatment process.
8. Waste sludge and brine	Waste sludge and brine are	No brine production, but longer	Sludge and waste production

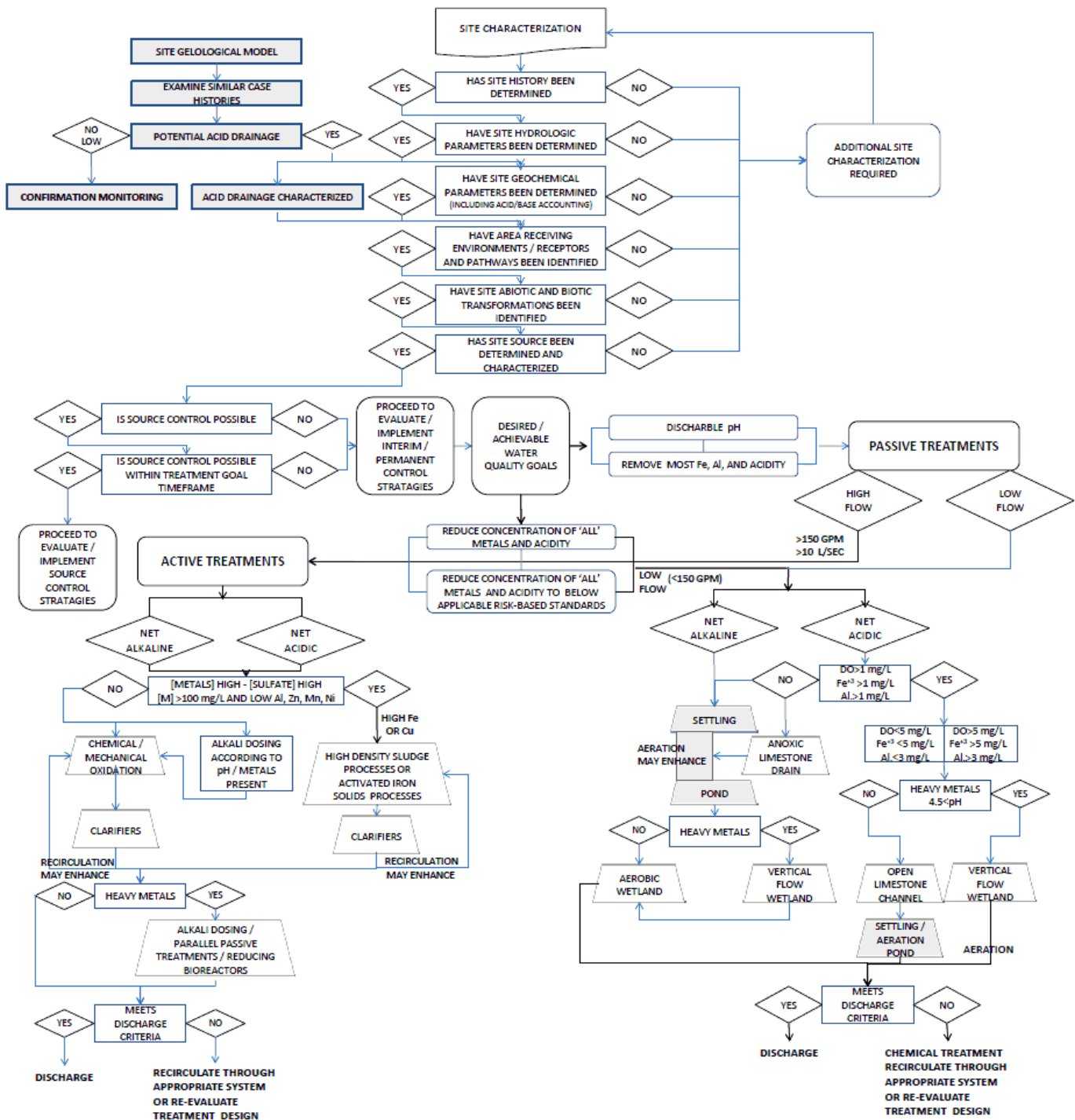
production.	produced, depending on level of treatment, requiring disposal.	term liability to deal with accumulated pollutants in wetland sludge.	accumulated in situ, may pose long term environmental liability.
9. Capital investment cost	High capital investment and periodic capital replacement required.	Moderate capital investment with periodic reinvestment to replace depleted wetlands media.	Low capital investment typically to deal with a short term problem.
10. Operating and maintenance cost	High operating and maintenance cost, with some potential for cost recovery by sale of product water, metals and by-products.	Low operating cost.	Moderate operating costs, but chemical usage may be high due to process inefficiency.

The costs of each ARD treatment system based on neutralization (in terms of the reagent amount and cost, capital investment, and maintenance of the dispensing system) and sludge disposal should be evaluated to determine the most cost effective system. The U.S. Office of Surface Mining has developed a software package, AMDTreat, which can be used to decide among the various options. AMDTreat can be downloaded at: <http://amd.osmre.gov/>. Where possible, users should apply local reagent prices rather than the default values. Another tool available is the Excel based ABATES program (<http://www.earthsystems.com.au/resources/acid-drainage/>) developed by Earth Systems for acid-base accounting and reagent requirements and treatment costs.

An excellent tool that can be used to decide which treatment process should be selected is the Acid Drainage Decision Tree (Figure 7-3) developed by Jack Adams (pers.comm.) of the University of Utah. Depending upon a range of factors, including influent flow rate, pH, acidity, alkalinity, the presence of iron, sulphate and other contaminants, the appropriate treatment process can be selected.

Figure 7-3: Acid Drainage Decision Tree

ACID DRAINAGE DECISION TREE (ACIDITY – SULFATE – Fe (II) – Al – Metals)



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7.5.1 Active Treatment Technologies

Active treatment refers to technologies requiring ongoing human operations; maintenance, and monitoring based on external sources of energy (electrical power) using infrastructure and engineered systems.

Active treatment technologies include aeration, neutralization, which often includes metal precipitation, metals removal, chemical precipitation, membrane processes, ion exchange, and biological sulphate removal.

7.5.1.1 Aeration

Since the principal contaminant is often dissolved ferrous iron, a key aspect of treating ARD is aeration. Only about 10 mg/L of oxygen can dissolve in water, so if there is more than about 50 mg/L of Fe₂⁺, the water must be aerated. Even at lower Fe₂⁺ concentrations, aeration increases the level of dissolved oxygen and

promotes oxidation of iron and manganese, increases chemical treatment efficiency, and decreases costs. Aeration also drives off dissolved CO₂, which is commonly present in mine water coming from underground. This increases the pH and can significantly reduce reagent use. To view photos of aeration systems, click here: [Aeration systems for treating CMD.](#)

Aeration can be done before or during treatment, using gravity or mechanical aeration/ mixing devices. In-line systems that use Venturi-based jet pumps and static mixers can be a cost-effective alternative since the air and neutralizing agent can both be introduced into the same jet pump orifice, increasing operational efficiency (Ackman and Kleinmann, 1984, 1991). If there is at least 20 psi (1.4 × 10⁵ Pa) of excess systemic water pressure (e.g., the water is being pumped to the treatment site), these simple in-line systems do not require additional power. Otherwise, a small amount of power can operate an aeration device.

The primary cost of aeration is in the blower power consumption. For example, a 40-HP blower operated 24 hours per day for a year will cost about \$18k/yr in power consumption at \$0.07/kWh. A second cost is in the mixing system, as proper aeration requires that a high-shear radial impeller be used to break-up the air bubbles and increase the surface area for oxygen dissolution. These mixers typically draw more than twice the power than would an axial agitator used solely to maintain the precipitates in suspension (Zinck and Aube, 2000).

A hidden cost is also included in the dissolution of carbon dioxide from air, which will increase lime consumption and sludge production. Although air contains only 0.03% carbon dioxide, the dissolution rate of CO₂ is considerably faster than that of oxygen. If aeration is not necessary, these additional lime costs and associated additional sludge disposal costs must also be considered (Zinck and Aube, 2000).

The capital costs of aeration include the purchase price of a blower, the air distribution system, and the radial agitator. Often a second blower is added as a backup, and either a separate building is constructed or a room is insulated for sound due to the high decibels put out by a blower.

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7.5.1.2 Neutralization/Hydrolysis

The key considerations in selecting an appropriate neutralization agent and integrated process configuration for a specific mine water treatment application include the following:

- Materials handling, including road/rail transport, bulk storage, make up, and dosing
- Classification of alkali material as a dangerous or hazardous material requiring special precautions in handling and personnel safety
- Availability and reliability of supply
- Efficiency as neutralizing agent and active ingredient/component of bulk material
- Process implications such as increasing propensity for scaling/coating/clogging of equipment /pipelines/instrumentation
- Infrastructure and equipment investment cost of alkali material handling, storage, make up, and dosing facilities

Neutralization and hydrolysis are key aspects of ARD treatment and many different alkali materials and different process configurations are employed. A list of commonly applied alkali compounds and materials is in Table 7-2.

Table 7-2: Alkali Materials and Compounds Applied to ARD Treatment

Alkali Compound/Material	Alkali Requirements (ton/ton of acidity) ¹	Neutralisation Efficiency (% of applied alkali used) ²	Relative Cost (\$ / tonnes bulk) ³
Limestone, CaCO ₃	1.00	30 - 50	10 – 15
Hydrated lime, Ca(OH) ₂	0.74	90	60 – 100
Un-hydrated (quick) lime, CaO	0.56	90	80 – 240
Soda ash, Na ₂ CO ₃	1.06	60 - 80	200 – 350
Caustic soda, NaOH	0.80	100	650 – 900
Magna lime, MgO	0.4	90	Project specific
Fly ash	Material specific	-	Project specific
Kiln dust	Material specific	-	Project specific
Slag	Material specific	-	Project specific

¹ The alkali requirement is expressed relative to CaCO₃ and reflects the amount of alkali required per unit of acidity (expressed as CaCO₃).

² Neutralization efficiency estimates the relative effectiveness of the chemical in neutralizing ARD acidity. For example, if 100 tons of acid was the amount of acid to be neutralized, then it can be estimated that 82 tons of hydrated lime would be needed to neutralize the acidity in the water (100(0.74)/0.90).

³ Price of chemical depends on the quantity being delivered. Bulk delivery prices and small quantity delivery prices will differ. These prices are approximate and generally reflect the market in January 2009. Prices will vary significantly around the world and over time.

Selection of an alkali material depends on the following:

- Secondary impacts associated with the use of a specific alkali residual on treated mine water quality such as ammonia content (aquatic environmental, eco-toxicity impacts), and increased salinity
- Cost of alkali material
- Treatment objectives, specifically the removal of metals

7.5.1.2.1 Lime

Hydrated lime ($\text{Ca}(\text{OH})_2$) is typically procured in bulk powder form. Lime can be added either as a controlled dispersion of powder into the water or as a lime slurry. Hydrated lime is particularly useful and cost effective in large-flow, high-acidity situations where a lime treatment plant with a mixer/aerator is constructed to help disperse and mix the chemical with the water (Skousen and Ziemkiewicz, 1996). Lime slurry piping requires careful design and maintenance due to the tendency of the lime to congeal in the piping system under certain conditions.

Lime neutralization in a high density sludge (HDS) process configuration is the industry standard for impacted mine water neutralization for of the following reasons:

- Relative low cost of lime
- Efficient use of lime
- High density of waste sludge requiring a smaller site for disposal
- Scale control on treatment plant structures, pipelines, equipment, and instrumentation
- Good solids/water separation
- Robust process, able to treat variable flows and acidity/metals loadings

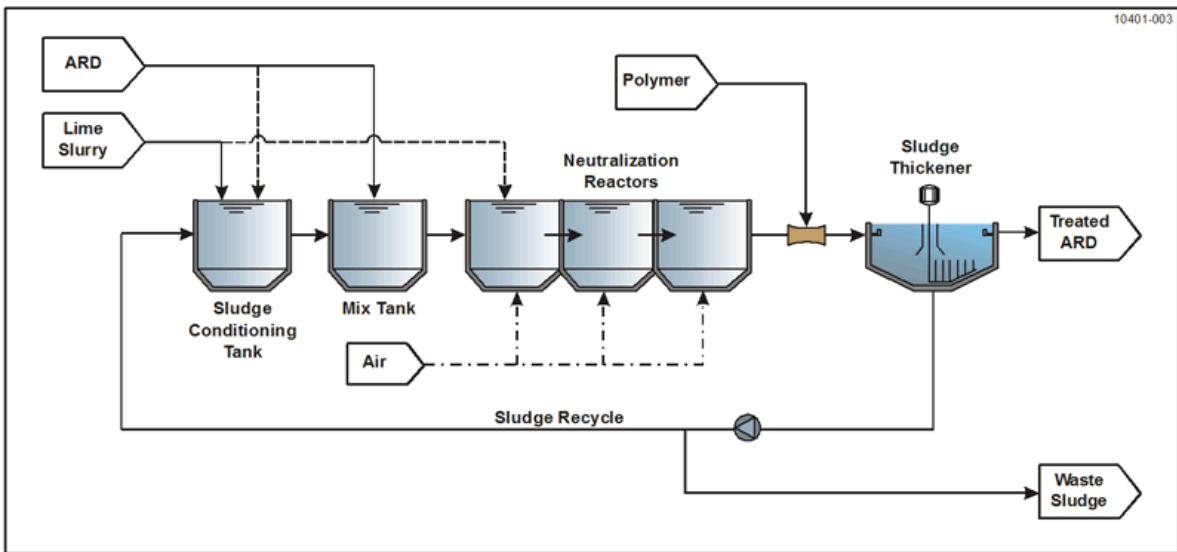
Lime neutralization/hydrolysis in an HDS process configuration is the most established and widely practiced ARD treatment technology. A number of variations and innovations to the original HDS treatment process concept have been developed and implemented. The basic HDS process configuration is shown in Table 7-3.

The key features of some of the commonly applied HDS process variations are shown in Figure 7-4.

Table 7-3: Comparative Table Different HDS Process Configurations

Process Parameters	Conventional HDS	Cominco Process	Geco Process	Staged-neutralization	Tetra (Doyon) Process
ARD feed point	Mix tank	Mix tank	Sludge conditioning tank	First stage	Sludge conditioning tank
Sludge recycle point	Sludge conditioning tank	Separate sludge/lime mix tank	Sludge conditioning tank	Upstream stages	Sludge conditioning tank and separate sludge/lime mix tank
Lime slurry feed point	Sludge conditioning tank	Separate sludge/lime mix tank	Rapid mix tank	Downstream stages	Separate sludge/lime mix tank
Aeration, air injection	Neutralization reactor	Neutralization reactor	Neutralization reactor	Upstream stages	Neutralization reactor
Polymer addition point	Upstream of thickener	Upstream of thickener	Upstream of thickener	Upstream of thickener	Upstream of thickener
Solids separation device	Gravity thickener	Gravity thickener	Gravity thickener	Gravity thickener	Gravity thickener

Figure 7-4: Basic HDS Process Configuration



The selection of the most appropriate lime neutralization process is site and project specific and will depend on the following:

- Flow rate and acidity/metals loadings
- Efficiency of lime usage
- Sludge settling and solid/liquid separation characteristics
- Waste sludge density and disposal site size (volume) constraints
- Sludge stability (residual neutralization capacity)
- Treated water quality
- Capital investment
- Operating and maintenance cost

Table 7-4 lists the relative performance of some lime neutralization processes based on a few selection criteria.

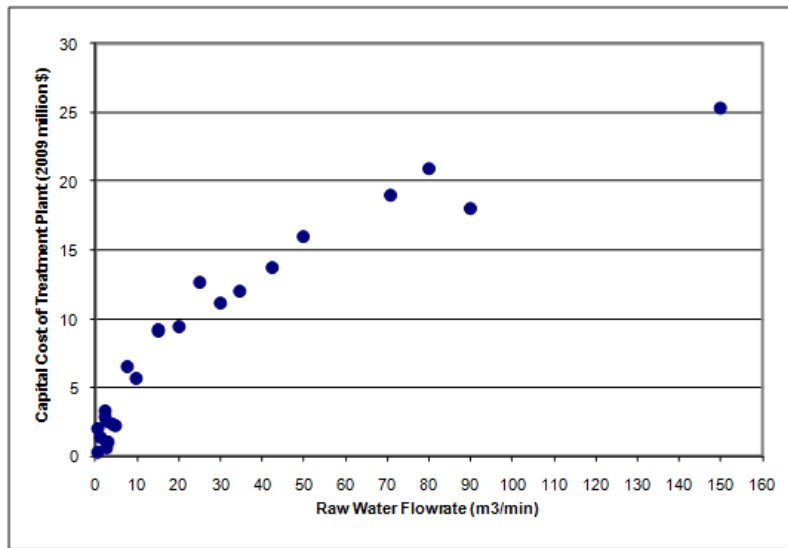
Table 7-4: Selection Criteria for Lime Neutralization Processes

Selection Criteria	Conventional HDS	Cominco Process	Geco Process	Tetra (Doyon) Process	Staged-neutralization
Efficient lime utilization	X	XX	XX	XX	XXX
Waste sludge density	X	XX	XX	XX	XXX
Sludge viscosity	XXX	XX	XX	XX	X
Sludge stability	XXX	XX	XX	XX	X
Treated water quality	XX	XX	XX	XX	XX
Legend	X	= good			
	XX	= better			
	XXX	= best			

The process principles for the Geco, Tetra, and Staged Neutralization treatment processes are similar and based on intermediate and final pH adjustment. Staged neutralization is better suited to the treatment of ARD with a high iron and sulphate content.

The capital costs of the treatment process are directly dependent on influent flow rate. Figure 7-5 shows the relationship between flow rate and capital cost based on actual treatment plant capital costs (Aubé, 2011)

Figure 7-5: Relationship between Flow Rate and Treatment Plant Capital Cost



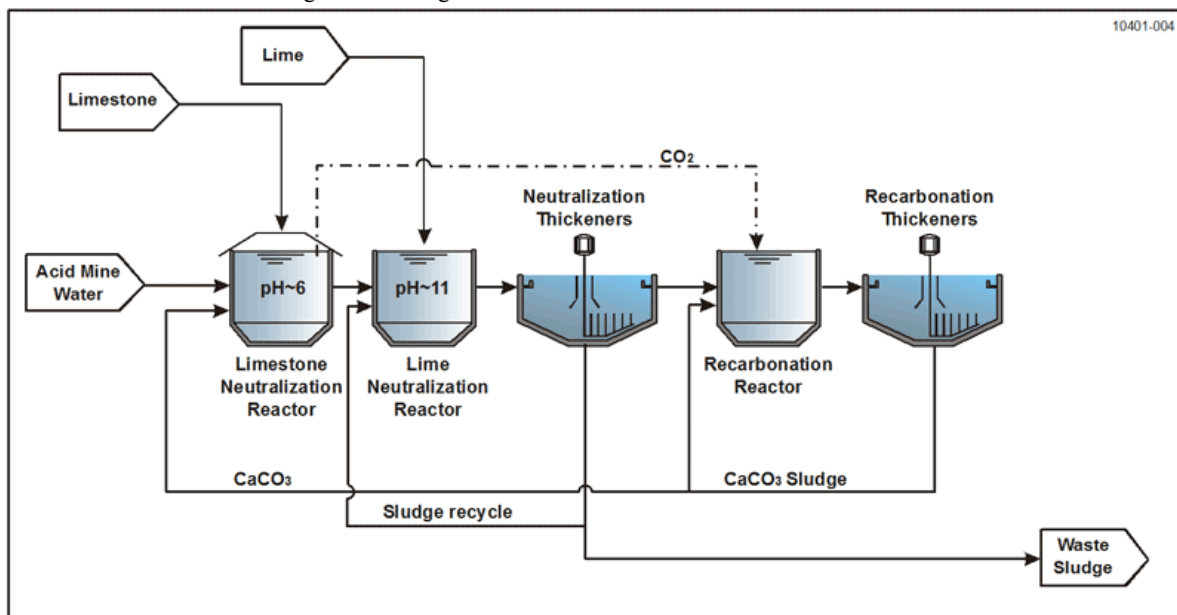
Hydrated lime is less cost effective if a very high pH is required to remove ions such as Mn, which is a common problem in CMD. Operators of lime treatment systems often increase lime application as Mn levels increase in the water. However, due to the kinetics of lime dissolution, increasing the lime rate increases the volume of unreacted lime that enters the metal flocc-settling pond. An additional complication is that it is relatively easy to over-treat CMD with hydrated lime, which can result in a pH that is high enough to cause aluminum to redissolve. If the treated water still has a pH above 9 once the iron hydrolyzes and settles, regulatory authorities will typically insist that the alkaline water be re-acidified to at least pH 9 unless the receiving stream is acidic. Using an in-line system (mentioned in Section 7.5.1.1), may allow an operator to better regulate lime usage in such circumstances. An example of an HDS lime neutralization system is provided in the Brukunga Case Study.

7.5.1.2.2 Limestone

Limestone has been used for decades to raise pH and precipitate metals in CMD (Deul and Mihok, 1967; Mihok, 1970). It has the lowest material cost and is the safest and easiest to handle of the ARD chemicals. It is useful when the only contaminants of concern are iron and aluminum, as is often the case in CMD. Unfortunately, its successful application is limited due to its low solubility and tendency to develop an external coating, or armour, of Fe(OH)₃ when added to ARD. Limestone, when simply placed into mine water, should be very fine grained (a high particle surface area/ volume ratio). The goal is for the limestone to dissolve before it becomes armoured. When pH is low and the metal concentrations are also relatively low, finely-ground limestone may be dumped into drainage directly (limestone sand application) or limestone gravel may be ground into powder by water-powered rotating drums (limestone drum stations) and metered into the drainage. Sand-sized limestone has also been placed in a large cylindrical tank and mixed with the ARD which is introduced into the bottom of the tank; these are called diversion wells (Faulkner and Skousen, 1995; Arnold, 1991). Diversion wells use the power of the drainage to fluidize (form a suspension) the limestone. The limestone particles rub against each vigorously, which allows dissolution without armoring. Limestone has also been used to treat CMD in anaerobic (e.g., anoxic limestone drains) and aerobic environments (e.g., open limestone channels). These are covered in more detail later in this chapter, as part of passive treatment.

A novel integrated limestone/lime neutralization process was developed at the South African Council for Scientific and Industrial Research (CSIR) (Geldenhuys et al., 2001), as shown in Figure 7-6. The integrated limestone/lime process incorporates the following three process steps:

Figure 7-6: Integrated Limestone / Lime Neutralization Process



- Pre-neutralization using relatively inexpensive limestone
- Lime neutralization to a pH target, which is dictated by the treatment targets such as specific metals removal (This step is also designed to precipitate

gypsum.)

- Re-carbonation and pH adjustment using the CO₂ generated in the first process step

The benefits of the integrated limestone/lime process relate to the efficient use of relatively inexpensive alkali materials and reuse of alkali sludge produced in the process.

Many process streams within mineral processing facilities are highly alkaline (i.e., waters from flotation plants). Therefore, excess process waters from the flotation plant could be mixed with ARD for neutralization.

7.5.1.2.3 Other Forms of Alkali Addition

At sites where it is not possible to provide as much supervision as the use of hydrated lime requires, pebble quicklime (CaO) has often been used, in conjunction with the Aquafix water treatment system. This device is powered by the force of the stream, using a water wheel concept (Jenkins and Skousen, 2001). The advantage of this approach is the amount of maintenance and manpower is much less than is required for a hydrated lime treatment plant, though it is still greater than is required for a passive treatment system. The amount of chemical added is dictated by the movement of a water wheel, which causes a screw feeder to dispense the chemical. The hopper and feeder can be installed in less than an hour. This system was initially used for small and/or periodic flows with high acidity because CaO reacts very quickly. Recently, water wheels have been attached to large silos for relatively high flow/ high acidity situations.

Caustic soda (NaOH) is available commercially as a concentrated liquid or as water soluble pellets. It is generally only used in remote locations (e.g., where electricity is unavailable), and in low flow, high acidity situations, especially where long- term ARD treatment may not be necessary or where Mn concentrations are high. Caustic soda is very soluble in water, disperses rapidly, and raises the pH of the water quickly. It should be applied at the surface of ponds because the chemical is denser than water. The major drawbacks of liquid NaOH for CMD treatment are its relatively high cost. Liquid NaOH is also extremely caustic and therefore potentially dangerous to anyone who comes in contact with it. A third drawback, at least in winter, is that it has a relatively high freezing point ($\approx 14^{\circ}\text{C}$), which has caused problems at some sites.

Soda ash (Na₂CO₃) is generally only used to treat CMD in remote areas with low flow and low amounts of acidity and metals. Selection of Na₂CO₃ for treating ARD is usually based on convenience rather than chemical cost. Soda ash comes as solid briquettes, and is gravity fed into water by the use of bins or barrels. The number of briquettes to be used each day is determined by the rate of flow and quality of the water being treated. One problem with the bin system is that the briquettes absorb moisture, causing them to expand and stick to the corners of the bin. This prevents the briquettes from dropping into the stream. For short-term treatment at isolated sites, some operators use a much simpler system, employing a wooden box or barrel with holes that allows water inflow and outflow. The operator simply fills the barrel with briquettes on a regular basis and places the barrel in the flowing water. However, this system offers less control of the amount of chemical used.

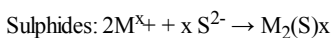
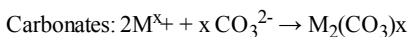
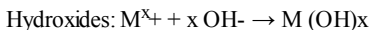
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7.5.1.3 Metal Removal

As discussed in Chapter 2, the metals content of mine drainage varies significantly depending on the following:

- Geology and geochemistry of the mine environment
- Specific ore being mined
- pH and oxidation/reduction potential of the mine water which governs the solubility of metals
- Source of mine water (e.g., drainage from underground workings, runoff from open pit workings, seepage from waste rock dumps, drainage from mill tailings and ore stock piles, spent ore piles from heap leach operations)
- Climatic conditions

The classical approach to metals removal is based on chemical precipitation, formation of solids particles containing the metal precipitates, and separation of the solids from the mine drainage. Metals [M] can form a number of insoluble compounds with anions, such as:



The solubility of metal hydroxides can be used to illustrate the point. Many metals have an amphoteric property, with decreasing solubility up to a threshold pH, above which the metal solubility increases again because of the formation of soluble complexes. The pH corresponding to the theoretical thermodynamic and minimum solubility of some selected metal hydroxides is shown in Table 7-5.

Table 7-5: Theoretical Minimum Metal Hydroxide Solubility pH

Metal	pH Corresponding to Minimum Metal Hydroxide Solubility/L
Ferric iron, Fe ³⁺	~ 3.5
Antimony, Sb ²⁺	~ 4.2
Aluminum, Al ³⁺	~ 4.5
Lead, Pb ²⁺	~ 6.5
	~ 7.0

Copper, Cu ²⁺	
Ferrous iron, Fe ²⁺	~ 8.0
Zinc, Zn ²⁺	~ 8.5
Nickel, Ni ²⁺	~ 9.3
Cadmium, Cd ²⁺	~ 10.0
Manganese, Mn ²⁺	~ 10.6

Metals removal by precipitation typically involves alkali addition to a target pH for selective removal of the metal of interest. It may also be advisable to pre-oxidize the metal (or metals) before precipitation where a metal can exist in more than one oxidation state. This will assist precipitation because the more oxidized form of some metals has a lower solubility. This is, however, not true for compounds of chromium, selenium, and uranium, which are more soluble in oxidized form.

A common approach to enhance removal of specific metals is the use of chemical pre-treatment or co-precipitation strategies, such as the following:

- Aeration can be used to improve removal of iron and manganese
- In low-iron containing waters, iron may be added to co-precipitate or adsorb certain metals onto ferric hydroxide precipitates. This process achieves lower effluent concentrations than would be achieved solely based on the solubility of the pure metal hydroxide
- Chemical reduction or oxidation can be used to alter the valence state of a target metal and enhance its removal. Examples of chemical reduction or oxidation include arsenic, selenium, and chromium

The key considerations in selecting an appropriate reagent for metal precipitation include:

- Materials handling considerations, including road/rail transport, bulk storage, make up, and dosing
- Classification of the reagent as a dangerous or hazardous material requiring special precautions in handling and personnel safety
- Availability and reliability of the supply
- Infrastructure and equipment investment cost of reagent handling, storage, make up, and dosing facilities
- Cost of reagent
- Treatment objectives

The specific process arrangement for metals removal is the same as for neutralization – and is often in a lime/HDS configuration with additional chemical feed and control systems. The primary differences are the potential pre-treatment requirements, operation at an elevated pH, and the possible need to reduce the treated effluent pH with acid or carbon dioxide to meet effluent discharge pH requirements.

After chemical treatment, the treated water will typically be directed into sedimentation ponds or mechanical thickeners so that the precipitating metals suspended in the water can precipitate and settle out. The metals generally precipitate from impacted water as a loose, open-structured mass of tiny grains called ‘floc’, which aggregates and settles out as a yellow, orange, or red sludge. Since CMD normally contains little in the way of potentially toxic contaminants, this sludge is generally non-hazardous, with the exception of those higher selenium contents. However, CMD sludge cannot be allowed to flow into the receiving stream since it would make the bottom of the stream inhospitable to fish, insect larvae and other benthic organisms. The sludge in the settling ponds must be periodically pumped out and disposed of, since sufficient residence time, which is dictated by pond size and depth, is important for adequate metal precipitation. The amount of metal floc generated by neutralization depends on the quality and quantity of water being treated, which in turn determines how often the ponds must be cleaned.

Sludge disposal options include: (1) leaving the material submerged in a pond indefinitely; (2) pumping or hauling sludge from ponds to abandoned deep mines or to pits dug on surface mines, or simply placing it onto the land surface; and (3) dumping sludge into refuse piles. CMD sludge is often disposed in abandoned deep mines or to pits dug on surface mines to take advantage of its excess alkalinity (due to unconsumed hydrated lime) but this is only appropriate if the environment that the sludge is being placed into is not acidic. If the sludge is exposed to sufficiently acidic water, the sludge can dissolve, neutralizing the pH somewhat but increasing the dissolved metal content. Sludge dewatering can be a cost effective alternative when the alternative is pumping or trucking sludge that is 80-95% water. CMD sludge pumped onto the surface of land and allowed to age and dry is generally a good strategy for disposal, since, in its oxidized and dried condition, the sludge can become crystalline and part of the soil. ARD sludge has also been dewatered and contained using geotextile products.

Selenium content and potential leaching from treatment sludge is in general not an issue as Se in its oxidized form does not readily report to the sludge. In a MEND study which examined seventeen treatment sludges from coal, base metal, precious metal, and uranium treatment operations, the concentration of Se leached was below regulated limits for all samples tested (Zinck et al., 1997).

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7.5.1.4 Chemical Precipitation for Sulphate Removal

The desalination treatment technologies most applicable to mine drainage target sulphate salts. Mine water may contain a wide range of anionic species, but sulphate is typical of many mine drainages and often represents the primary contaminant. Consequently, sulphate removal is an important treatment objective and is also often key to the reduction of TDS.

Some sulphate is removed by gypsum precipitation during neutralization reactions if lime, limestone, or another calcium source is added during water treatment. In addition, a number of precipitation processes have been developed for specific application to high sulphate content mine waters, including the following:

- Barium sulphate process
- Ettringite (Ca₆Al₂(SO₄)₃(OH)₁₂•26H₂O) precipitation process

The barium sulphate process is based on the addition of a barium salt to re-precipitate sulphate. The insoluble barium sulphate sludge is separated and removed from

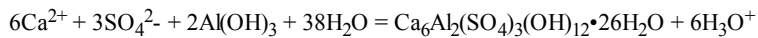
the main stream process. This barium is recovered from the sulphate sludge and recycled to the main stream process.

The barium sulphate process has not been developed past the development of a pilot scale demonstration process. While this process is very effective, it is challenged by the following:

- The use of an environmentally toxic compound as a treatment reagent
- Generation and handling of a toxic and hazardous gas (H₂S)
- Requirement for thermal regeneration and recycle of the barium reagent

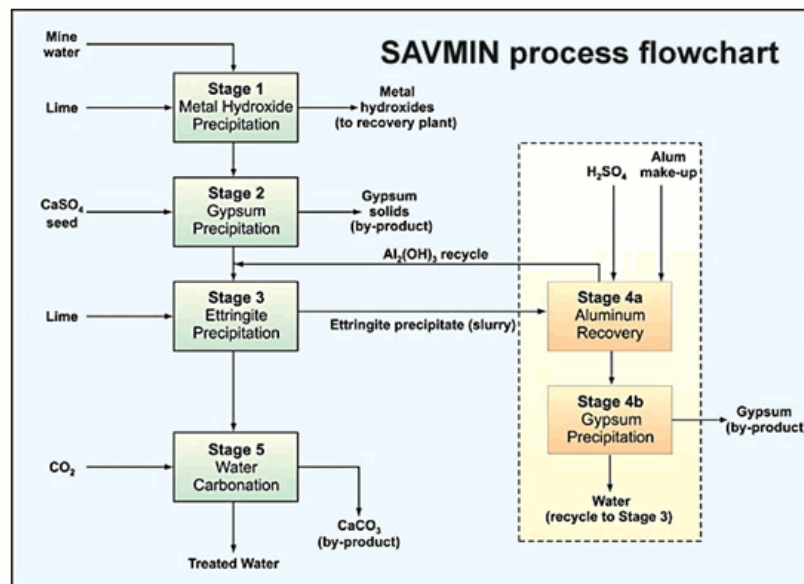
Barium carbonate and barium hydroxide have been tested by CANMET-MMSL in Canada. (Zinck et al., 2007).

Two variations of the ettringite precipitation process (SAVMINTM and cost-effective sulphate removal [CESR]) have been developed and demonstrated. The ettringite process is based on the addition of aluminum hydroxide in a high pH environment resulting in precipitation of ettringite (a hydrated calcium aluminosulphate mineral), as shown below:



The simplified process flow diagram of the SAVMINTM process is shown in Figure 7-7.

Figure 7-7: Simplified SAVMIN Process Diagram



The CESR process is similar in concept but CESR uses a proprietary chemical derived from the cement industry to precipitate ettringite. It has the benefit of not requiring the decomposition of ettringite or the recycling of reagents.

While these processes have been demonstrated, neither has been applied to mine drainage projects for full-scale installations to date.

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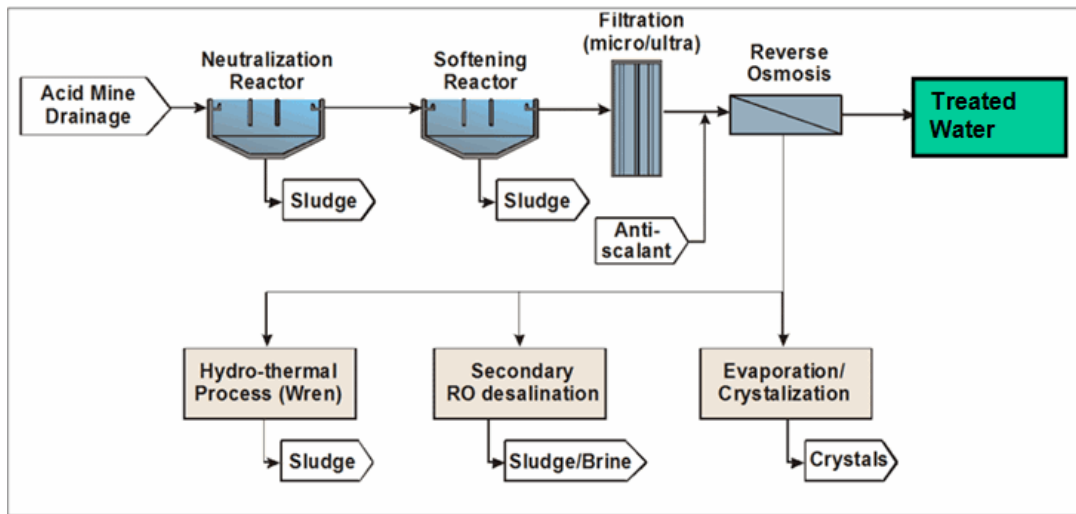
7.5.1.5 Membrane Treatment

A wide range of membrane treatment technologies exist to treat brackish and saline waters such as mine drainage. The application of these membrane technologies to mine drainage is challenging because of scaling and fouling potential. Mine drainage typically contains several compounds with a scaling and fouling potential such as metals, sulphate, and carbonate. The application of membrane desalination processes to mine drainage also typically results in the production of sludge and brine streams. In recent years, however, a number of high recovery membrane desalination processes have been developed, constructed, and operated at mine sites.

The concept of a high recovery membrane desalination process is shown in Figure 7-7. The primary features of the mainstream membrane desalination process include the following:

- Pretreatment with lime to remove metals and supersaturated gypsum (this is essential to limit the membrane scaling potential of the mine drainage)
- Pretreatment to remove residual suspended solids
- Pretreatment by adjusting the pH to a nonscaling regime and adding anti-scalant reagent
- Membrane treatment typically accomplished using spiral wound reverse osmosis (RO) or nano-filtration (NF) membranes
- Post treatment (a simple process that may only involve stabilization using an alkali such as lime)

Figure 7-8: Conceptual High Recovery Membrane Desalination Process

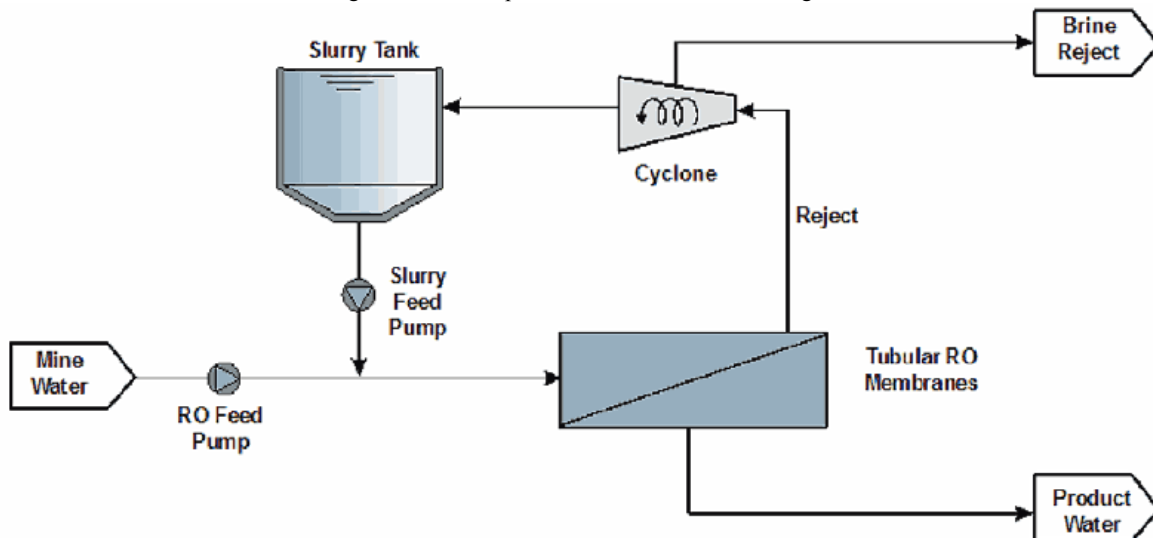


A single pass membrane treatment process will typically achieve only a clean water recovery of 60% to 70% for mine waters. The membrane process still leaves a substantial brine stream, which requires treatment. Methods to treat the brine are discussed in Section 7.6. The following two approaches exist to further increase the clean water recovery and decrease the need for brine handling and disposal:

- The brine stream can be desaturated by lime treatment which destroys the anti-scalant action and precipitates any supersaturated salts. A second stage higher pressure RO/NF process is then used to recover more clean water.
- The brine stream can be further concentrated by conventional thermal evaporation/crystallization treatment. These techniques are capital intensive and require substantial energy.

A further variation of the membrane desalination process involves the use of tubular RO type membranes. The slurry precipitation and recycle reverse osmosis (SPARRO) process was developed and holds potential as shown in Figure 7-9.

Figure 7-9: Concept SPARRO Process Flow Diagram



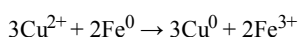
The concept of the SPARRO process is based on the protection of the membrane surfaces by providing a slurry suspension onto which the precipitation products can form. High water recoveries were achieved by a demonstration scale plant (Pulles et al., 1992).

In principle, other membrane processes such as electrodialysis reversal (EDR) can also be applied to mine water desalination. No full-scale EDR desalination plants, however, are known to exist in the mine water industry for the large-scale desalination of mine drainage.

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7.5.1.6 Ion Exchange

One of the older ion exchange processes used by mining companies is the copper cementation or precipitation process. In this process, waste galvanized cans were burnt to remove the zinc coating or other metallic iron was placed in a copper containing stream, which was typically leach solution from a waste or low-grade ore pile. Copper in solution would plate on the surface of the iron metal and in doing so would exchange electrons with the underlying iron, oxidizing the iron and reducing the copper to the metallic state. This process created a higher-value product from a waste product (precipitate copper from waste cans) and would reduce somewhat the toxicity of the solution to fish (exchanging Cu^{2+} for Fe^{3+} ions), as shown in the following reaction:



This process has been used to treat copper containing solutions from abandoned mine sites in the United States and it is a process that is amenable to use by artisan miners in developing countries, provided the influent copper concentration is quite elevated (i.e., greater than approximately 20 mg/L).

A novel ion exchange process, GYPCIX®, was developed for high sulphate type mine drainage. The process requires pretreatment to remove metals, which may interfere and decrease the efficiency of the downstream ion exchange process resins. The GYPCIX® conceptual flow diagram is shown in Figure 7-10. The cation resin exchanges Ca^{2+} , Mg^{2+} , and other cations (i.e., metal ions) by the following reaction:



The anion resin exchanges SO_4^{2-} , Cl^- , and other anions by the following reaction:

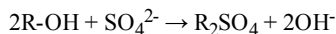
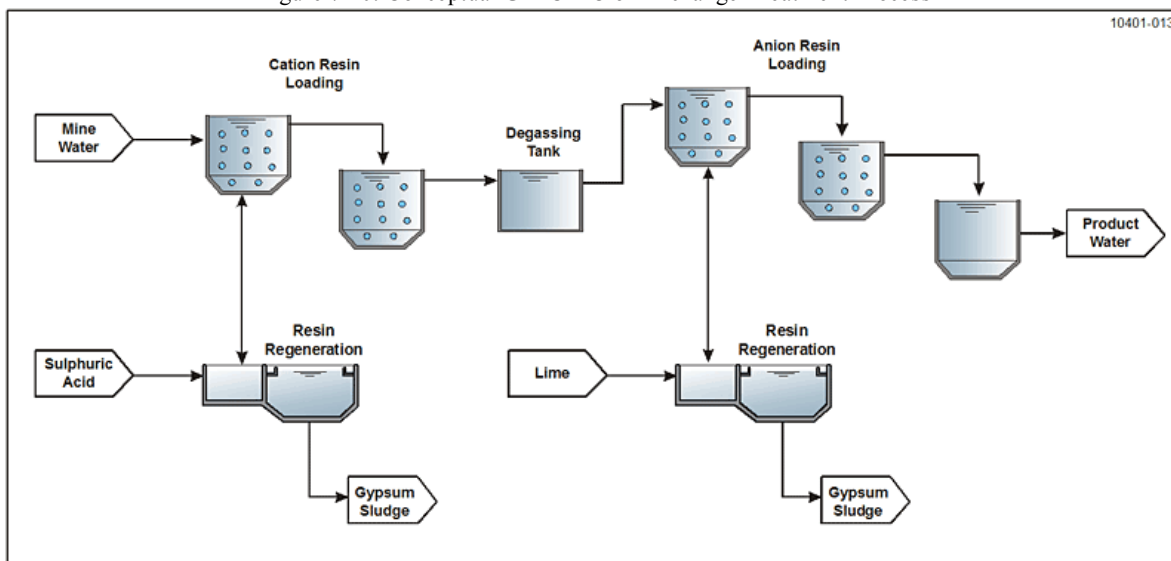


Figure 7-10: Conceptual GYPCIX® ion Exchange Treatment Process



The product water is near neutral and may require stabilization before distribution or discharge. The resin regeneration requires sulphuric acid and lime, thus producing mainly gypsum as waste sludge. The GYPCIX process has been demonstrated on a small scale, but no commercial operations exist in the mining industry.

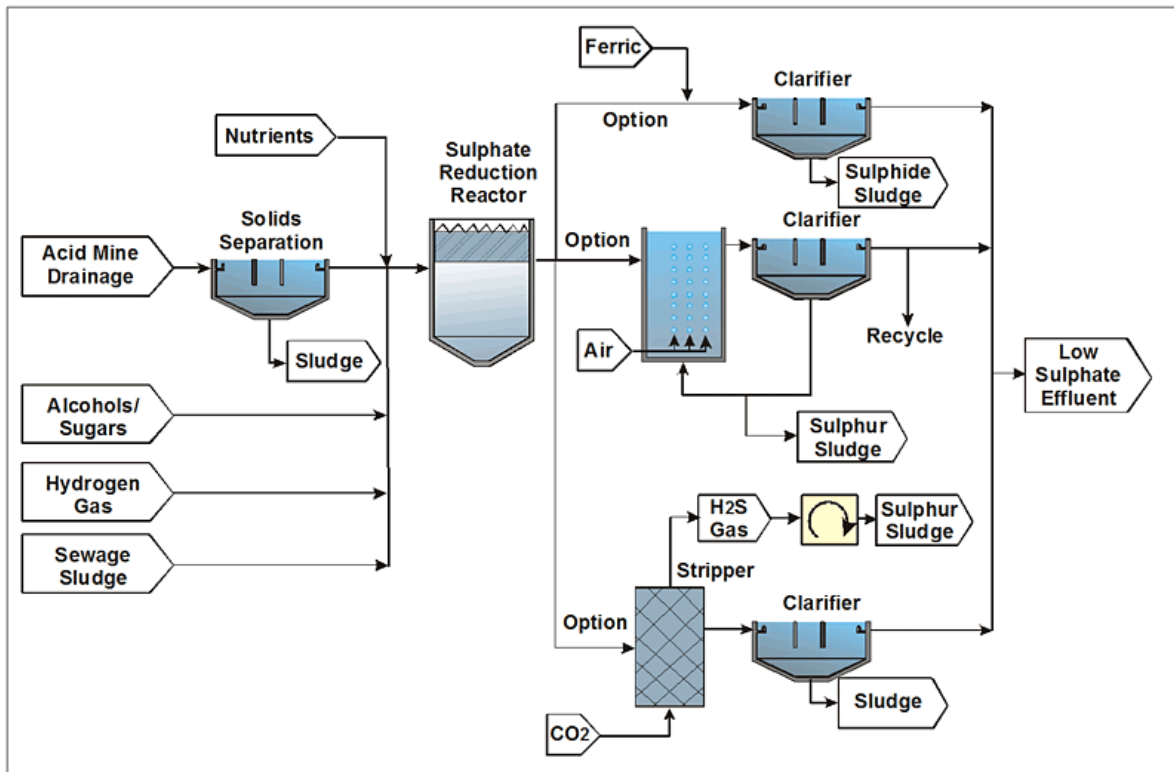
A number of natural ion-exchange materials, such as zeolites (a class of aluminosilicate minerals), have been demonstrated to have treatment potential. Few full-scale operating treatment facilities using natural ion-exchange materials exist.

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7.5.1.7 Biological Sulphate Removal

Biological sulphate removal has been used by mining companies at several locations around the world. Many variations of the process have been developed. The generic biological sulphate removal process configuration is shown in Figure 7-11.

Figure 7-11: Generic Biological Sulphate Removal Process Configuration



The key features of the biological sulphate removal process include the following:

- Pretreatment to remove metals by precipitation as sulphides, hydroxides, or carbonates
- Dosing of an electron donor and carbon source such as alcohol, sugar, H₂ gas, and even complex substrates such as sewage sludge
- Addition of nutrients, including sources of nitrogen, phosphate, potassium, and trace minerals
- Sulphate reduction in an anaerobic reactor which converts sulphate to sulphide. The process is mediated by sulphate reducing bacteria (SRB), which uses preferred substrates such as fatty acids, alcohols, and H₂ gas. The bacterial population includes a consortium of other organisms such as fermenting bacteria and methanogens, some of which help to hydrolyze and ferment complex carbons to readily available substrates for the SRBs

The biological sulphate reduction part of the process has been researched and demonstrated by a number of companies. This part of the overall treatment train can be considered as proven technology. The further handling and treatment of the sulphide rich effluent can be done in a number of different ways, as shown in Figure 7-11. A ferric salt (or ferric sludge) can be dosed to precipitate the sulphide; a ferric sulphide sludge is then generated, which may require special care in disposal and the associated anion may increase salinity of the treated water, as follows:

- The sulphide can be partially oxidized to sulphur in a carefully controlled micro-aerobic environment. The sulphur is separated as a potentially saleable by-product
- The sulphide is stripped and converted to sulphur in a side stream process. The substitution of H₂S by CO₂ results in an increase in carbonate alkalinity and potential precipitation of carbonates such as calcite

The criteria for selecting an appropriate mine water desalination technology are listed in Table 7-6, with an indication of the relative performance of different technologies.

Table 7-6: Criteria for Selecting an Appropriate Mine-Water Treatment Desalination Technology

Selection Criteria	Mine Drainage Treatment Technology			
	Chemical Precipitation	Membrane Treatment	Ion Exchange	Biological Sulphate Removal
Proven technology on commercial scale	Proven with many demonstration scales, large commercial plants	Proven, with several large commercial plants	Demonstrated on pilot scale, no large commercial plants	Proven, with a limited number of commercial plants
Specialized application	General application to high metals, high SO ₄ mine water	General application, but with appropriate pre-treatment	Demonstrated for CaSO ₄ type waters, with appropriate pre-treatment	Specialized application to high SO ₄ mine waters
Water recovery	High water recovery > 95%	High water recovery > 90%	High water recovery not confirmed	Very high water recovery > 98%

Waste sludge/brine production	Large waste sludge production	Sludge and brine production	Large waste sludge production	Small waste sludge production
Potential byproducts recovery	Potential for CaSO ₄ recovery	Potential, but not demonstrated	Potential for CaSO ₄ recovery	High potential for Sulphur recovery
Chemicals dosing	High chemicals dosing	Limited chemicals dosing	High chemicals dosing	Process depends on carbon source dosing
Energy usage efficiency	Moderate energy usage	High energy usage	Moderate energy usage	Moderate energy usage (heating of anaerobic reactors)
Reliable and robust performance	Robust process	Process good performance, but sensitive to pre-treatment	IX process performance and resin recovery subject to interference	Biological process sensitive to toxics, fluctuating feed water quality and environmental conditions
Capital investment cost (per m ³ /day capacity)	\$ 300 – 1,250 (see note)	\$500 – 1,000	See note	\$800 – 1,500
Operations and maintenance cost (\$ per m ³ treated)	\$0.2 – 1.5/m ³ (see note)	\$0.5 – 1.0/m ³	See note	\$0.7 – 1.5

Note: The cost information on chemical precipitation and ion exchange processes is indicative since no full scale commercial installations exist.

The cost of treating mine waters in cold and remote sites (i.e., arctic regions) could be higher by a factor of 2 or more because of expensive transportation and storage requirements of reagents. These sites are usually accessed by air or ice roads and the treatment systems are installed and operated indoors, requiring construction and maintenance of heated buildings. At closed mining sites, possibilities for seasonal operations should be investigated and applied where possible.

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7.5.1.8 Sulphide Precipitation

Sulphide precipitation works under the same basic principle as hydroxide precipitation. The precipitation process converts soluble metal compounds into relatively insoluble sulphide compounds through the addition of precipitating agents, such as the following:

- Sodium sulphide (Na₂S)
- Sodium hydrosulphide (NaHS)
- Ferrous sulphide (FeS)
- Calcium sulphide (CaS)

Sulphide precipitation is an effective alternative to hydroxide precipitation. Over a broad pH range, sulphides (S²⁻, HS⁻) are extremely reactive with heavy metal ions. Sulphide precipitation can be used to remove lead, copper, chromium (VI), silver, cadmium, zinc, mercury, nickel, thallium, antimony, and vanadium from wastewaters. The precipitation reaction is generally induced under near neutral conditions (pH 7.0 to 9.0). In a way that is similar to hydroxide precipitation, metal-sulphide precipitates must be physically removed from solution (through coagulation, flocculation, and clarification, or filtration), leaving a metal-sulphide sludge.

In addition, sulphide precipitation is sometimes used in water treatment following conventional lime treatment to reduce concentrations of residual metals, particularly cadmium. This is successful because of the ability of sulphide to reduce metal concentrations to much lower values than can be achieved by precipitating metals as hydroxides with lime, although the metals precipitated are not recovered as they report to the lime sludge. Some of the advantages of sulphide treatment include effective metal removal for most metals, low retention time requirement, and reduced sludge volumes. The disadvantages of sulphide treatment are significant and include potential for toxic hydrogen sulphide gas emissions and residual sulphide in treatment effluent. Also, the soluble sulphide process may result in odour problems and the complexities of the systems frequently result in higher capital and operating costs than lime treatment.

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7.5.2 Passive Treatment Technologies

Passive treatment refers to processes that do not require regular human intervention, operations, or maintenance. It should typically employ natural construction materials, (e.g., soils, clays, and broken rock), natural materials (e.g., plant residues such as straw, wood chips, manure, and compost) and promote the growth of natural vegetation. Passive treatment systems use gravity flow for water movement. In some arid climates, it might also include use of evaporation or infiltration (e.g., soil amelioration and neutralization) of small volumes of ARD.

Pulles et al. (2004) defined a passive treatment system as:

“A water treatment system that utilizes naturally available energy sources such as topographical gradient, microbial metabolic energy, photosynthesis and chemical energy and requires regular, but infrequent maintenance to operate successfully over its design life”

Gusek (2002) also defined passive treatment as:

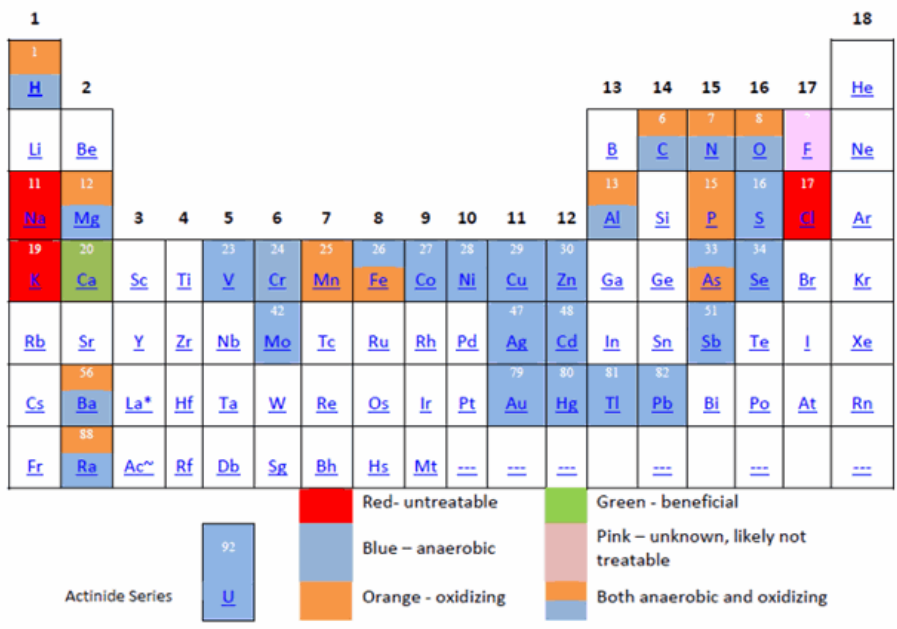
“... a process of sequentially removing metals and/or acidity in a natural-looking, man-made bio-system that capitalizes on ecological and geochemical reactions. The process does not require power or chemicals after construction, and lasts for decades with minimal human help”.

A truly passive system should also function for many years without a major retrofit to replenish materials, and should be able to function without using electrical power. Benning and Otte (1997) describe a volunteer passive system at an abandoned lead-zinc mine in Ireland that has apparently been functioning unattended for over 120 years. Similar volunteer systems are likely to be found functioning at some level of efficiency in most historical mining districts. Attempts to reproduce the beneficial effects observed at such volunteer wetlands in the eastern U.S. led to the use of passive treatment technology at mine sites.

Gusek (2008) provides an excellent summary of the history of passive treatment as applied to ARD and CMD in the US. The pioneering work of a group of researchers at Wright State University over thirty years ago documented water quality improvements in a natural Sphagnum bog in Ohio that was receiving low-pH, metal-laden water. Complementing this research, a group at West Virginia University found similar results at the Tub Run Bog. Subsequently, researchers, practitioners and engineers focused on developing the promising technology of using constructed wetlands to treat acid drainage. Since the term ‘wetland’, carried legal and regulatory challenges and does not quite describe structures like anoxic limestone drains or successive alkalinity producing systems, the term “passive treatment” was adopted. More detail on the history of passive treatment can be found here: **History of Passive Treatment**

Most elements can be treated in a passive treatment process as outlined in Figure 7-12, which is the periodic table for passive treatment elements developed by Gusek and Waples (2009).

Figure 7-12: Periodic Table for Passive Treatment



The generic categories of passive treatment systems are detailed in Table 7-7.

Table 7-7: Generic Categories of Passive Treatment Systems

Passive Treatment Technology	Application Niche in Mine Drainage
Aerobic wetlands	Net alkaline drainage
Anoxic limestone drains (ALD)	Net acidic, low Al ³⁺ , low Fe ³⁺ , low dissolved oxygen drainage
Anaerobic wetlands	Net acidic water with high metal content
Reducing and alkalinity producing systems (RAPS)	Net acidic water with high metal content
Open limestone drains (OLD)	Net acidic water with high metal content, low to moderate SO ₄ .

The proven application of passive treatment technology is to the low-flow range. Most successful passive treatment projects are treating less than 1,000 m³ per day. The largest documented passive treatment system has been treating approximately 6,500 m³ per day since 1996 with limited maintenance (Gusek et al., 2000, 2007).

Hedin et al. (1994a) developed a decision support flow sheet to assist in the selection of an appropriate passive treatment technology. This was further refined by the PIRAMID Consortium as shown in Figure 7-13 (PIRAMID Consortium, 2003). Gusek (2008) further updated the decision tree to include a wider range of chemistries for mining-influenced water as earlier versions primarily focused on iron and magnesium (Figure 7-14).

Figure 7-13: Selection of Passive Treatment Technology Chart

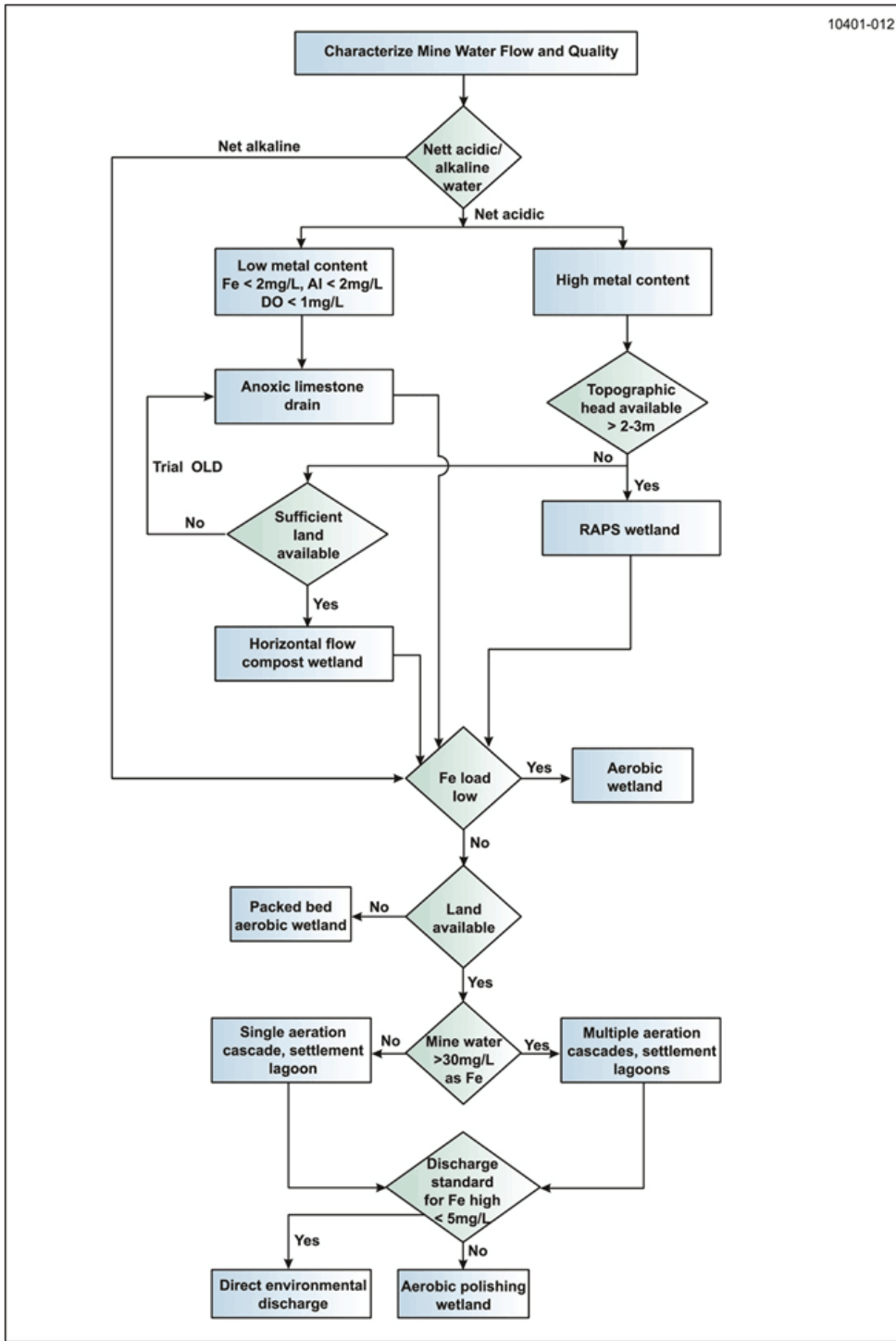
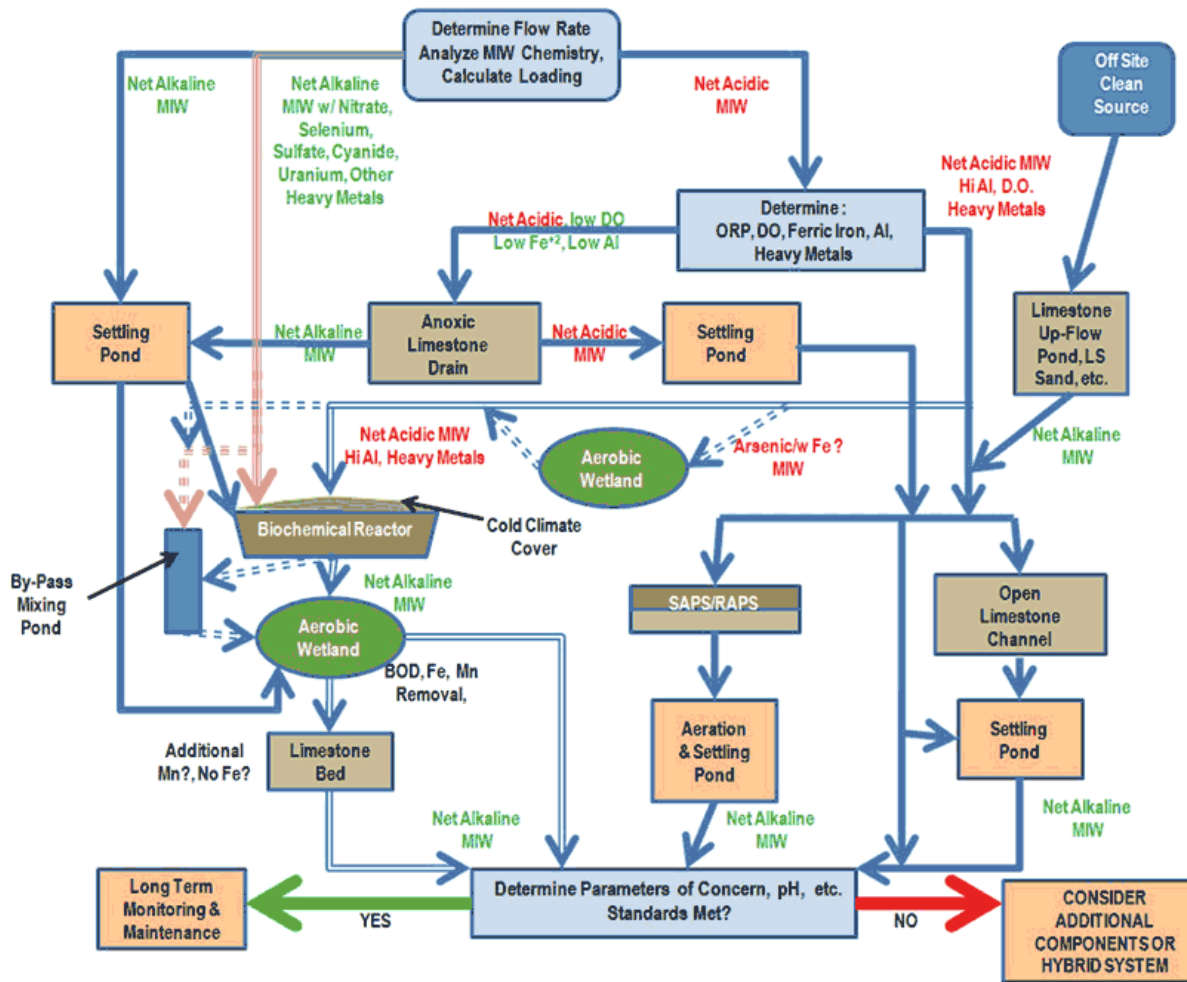


Figure 7-14: Selection of Passive Treatment Technology Chart



The mechanisms of metal removal and retention in passive treatment systems are varied and include:









- Oxidation
- Precipitation as hydroxides and carbonates under aerobic conditions
- Precipitation as sulphides and hydroxy-sulphate (aluminum special case) under anaerobic conditions
- Complexation and adsorption onto organic matter
- Ion exchange with organic matter
- Uptake by plants (phyto-remediation)

The environmental conditions in the different passive treatment systems will dictate the dominant metals removal mechanisms. Experience in Australia suggests that passive treatment is more effective if the acidity loading is less than approximately 150 kg/day.

The precipitation of iron as hydroxides and carbonates may also assist in the removal of additional pollutants. Several ionic species, such as arsenic and molybdenum, co-precipitate or adsorb onto ferric hydroxides. There is evidence that some of these reactions can be microbially facilitated (LeBlanc et al., 1996).

Sections 7.5.2.1 through 7.5.2.7 provide a brief overview of the principal passive treatment technologies. Figure 7-15 presents photographs of the main components of passive treatment systems (from Gusek, 2008).

Figure 7-15: Selected Passive Treatment Components (Gusek et al.)

Aerobic Wetland		Anoxic Limestone Drain photo credit: www.wvu.edu/~agexten/landrec/land	
Aeration and Settling Pond		Sulfate Reducing Bioreactor (AKA Biochemical Reactor – [BCR])	
Successive/ Reducing alkalinity producing systems (SAPS/ RAPS) photo credit: www.wvu.edu/~agexten/landrec/		Open limestone channels (OLCs) photo credit: www.wvu.edu/~agexten/landrec/	
Limestone beds		Limestone Sand (semi-passive)	

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7.5.2.1 Aerobic Wetlands

Aerobic wetlands provide the environmental conditions for removal of suspended solids and selected metals using the following features:

- Relatively shallow water depths to allow aeration of the mine drainage
- Cascades to further enhance aeration
- Configuration and layout to promote favourable hydrodynamic flow conditions (prevent short circuiting)
- Wetlands vegetation to assist in aeration of the substrate (wetlands vegetation has the capability to maintain aerobic conditions around the root/rhizome area and can also promote favourable flow conditions)
- Sufficient residence time to allow the treatment reactions to take place
- Space for the settling and accumulation of the metal precipitates and solids
- Layout and screening against wind mixing and re-suspension of settled solids
- Promote algal growth to further increase the pH and facilitate manganese oxidation and precipitation
- Piping and hydraulic controls to manage the water levels in individual wetlands cells

As mentioned previously, aeration can be enhanced passively by simply cascading the ARD down a rock-lined channel or over a dam to encourage splashing and turbulence. Mine impacted water that contains less than 50 mg/L of dissolved iron and relatively low concentrations of manganese can often be treated simply using this form of aeration, followed by a pond or wetland for metal floc settling, if there is enough change in elevation to produce the required turbulence. No chemical addition is needed and the water can be discharged safely without adversely affecting receiving streams. If the iron concentrations are higher, additional aeration steps can be incorporated into the design by inserting additional turbulence between the ponds or wetland cells. Since enhanced iron oxidation and hydrolysis are the keys to most ARD and CMD passive treatment systems, such turbulence steps are routinely added between wetland cells. For sites where the iron loading is particularly high, or where the change in elevation is minimal, supplemental aeration may be necessary. At such sites, semi-passive systems have been constructed using gravity-, wind-, and water-powered aeration or neutralization processes, as well as some devices that require external electrical power, may be used to provide supplemental aeration.

This is still often less expensive than conventional chemical treatment.

Aerobic wetlands consist of *Typha* and other wetland vegetation typically planted in shallow water depths (<30cm), in relatively impermeable sediments comprised of soil, clay, or mine spoil. Aerobic wetlands promote metal oxidation and hydrolysis, thereby causing precipitation and physical retention of Fe, Al, and Mn oxyhydroxides, much like sedimentation structures. Successful metal removal depends principally on the dissolved metal concentrations, dissolved oxygen (DO) content, pH, net acidity/alkalinity of the mine water, and the retention time of the water in the wetland. The pH and net acidity/alkalinity of the water are particularly important because pH influences both the solubility of metal hydroxide precipitates and the kinetics of metal oxidation and hydrolysis. Therefore, aerobic wetlands are best used in conjunction with water that is net alkaline; the wetland serves primarily as a metal-floc collection and retention structure. The wetlands must be designed properly to optimize sedimentation and to provide for sludge storage. The vegetation enhances physical filtration of suspended metal particles and colloids; direct metal uptake by the plants is usually only a significant factor when the metal concentrations are already very low.

Some aerobic systems have been constructed by planting *Typha* rhizomes in soil or alkaline spoil obtained onsite, while others have been planted simply by spreading *Typha* seeds, with good plant growth after two years. However, it is best to use a mixture of appropriate emergent vegetation since this will allow the wetland to survive better in times of stress. For the same reason, the wetland cells should not be of uniform depth, but should include shallow and deeper areas and a few deep (1 to 2 m) spots. Most rooted aquatic vegetation cannot tolerate water depths greater than 50 cm, and require shallower depths for propagation. However, varying the depth will help promote wetland diversity (with respect to both plants and animals) and will help the wetlands survive droughts and storm events.

Some of the aerobic systems that have been constructed to treat alkaline mine water have little emergent plant growth and are better termed ponds than wetlands. In fact, typically, since sludge can be pumped from a pond, a pond is typically placed before the wetland cells to remove much, if not most, of the iron hydroxide. This pond is usually sized for an 8 to 24 hour retention time (often encompassing as much surface area as the wetland cells that follow it) and is typically 1.5 to 2.5 m deep. To account for the accumulation of iron, the value of 0.17 g of iron per cm³ can be used, so that the required detention time will be available for a predetermined time (i.e. its design life). It is recommended that the freeboard of aerobic wetlands/ponds be constructed at about 1 m to allow for the removal of iron. Observations of sludge accumulation in existing wetlands suggest that a 1-m freeboard should be adequate to hold 20 to 25 years of iron oxyhydroxide accumulation. Some of these iron precipitates have been characterized for potential recycling as pigment (Kairies et al., 2001 and Hedin, 2002).

Often, several wetland cells and/or ponds are connected by flow through a v-notch weir, lined railroad tie steps, or down a ditch. Use of multiple cell/ponds can limit the amount of short-circuiting, and aerates the water at each connection. If there are elevation differences between the cells (as discussed above, to increase dissolved oxygen), the interconnection should be designed to dissipate kinetic energy and avoid erosion and/or the mobilization of precipitates in the next cell. Spillways should be designed to pass the maximum probable flow. Spillways should consist of wide cuts in the dike with side slopes no steeper than 2H:1V, be lined with non-biodegradable erosion control fabric and a coarse riprap, if high flows are expected (Brodie, 1991). Proper spillway design can preclude future maintenance costs associated with erosion and/or failed dikes. If pipes are used, small diameter (< 30 cm) pipes should be avoided because they can plug with litter and FeOOH deposits. Pipes should be made of PVC or PE, or coated for long-term stability. More details on the construction of aerobic wetland systems can be found in Hammer's *Creating Freshwater Wetlands* (1992). The floor of the wetland cell may be sloped up to a 3% grade. If a level cell floor is used, then the water level and flow will be controlled by the downstream dam spillway and/or adjustable riser pipes.

Hedin et al. (1994a) reported typical removal rates of 10 to 20 gd⁻¹m² for iron, and 0.5 to 1.0 gd⁻¹m² for manganese. Several groups have attempted to develop models that more effectively estimate the performance of treatment systems, especially for iron removal. Watzlaf et al. (2001, 2004) were able to model a system consisting of an aerobic pond, an aeration cascade, and a wetland using only the temperature-adjusted abiotic rate of iron oxidation. They found that the overall performance and the performance of certain sections of the system fell within the 10 to 20 gd⁻¹m² range, but that the performance of some sections was outside of that range. Their model indicated that pH was the key factor limiting the rate of removal.

Kirby et al. (1999) used the same factors but included the effect of bacterial iron oxidation to model a set of 17 ponds. They found that the relative importance of the biotic and abiotic mechanisms was determined mainly by pH, with the abiotic path predominating at the higher pH values. They suggested that pH and temperature are the most important variables for determining iron oxidation rates, and therefore, iron removal rates. However, little can be done to control temperature in a passive treatment. The work by Kirby et al. (1999) suggests that increasing pH from 6.1 to 6.4, for example, greatly enhances oxidation, whereas doubling dissolved oxygen (as long as oxygen is sufficiently high stoichiometrically to oxidize metals), pond volume, or retention time has considerably less impact on oxidation rates.

Dempsey et al. (2001) found that oxygen transfer was rate limiting in one system, and that the amount of catalytic reaction provided by ferric hydroxide was the determining factor at a second site. While heterogeneous catalysis apparently plays a significant role in iron oxidation, it is difficult to increase concentrations of iron solids in a completely passive system. Such catalysis could be quite important in semi-passive or active treatment systems.

However, overall, it appears that the original estimate of Hedin et al. (1994a) of 10 to 20 gd⁻¹m² remains a convenient pre-construction rule-of-thumb for estimating pond and wetlands sizes for iron removal. Studies undertaken since their publication tend to support the findings in the majority of cases (Younger et al., 2002 and Watzlaf et al., 2004). Recently, however, Kruse et al. (2009) suggested that hydraulic retention time rather than surface area should be used to design such systems.

The layout and slope of aerobic wetlands should be designed to minimize disruption of the natural conditions when the wetland sludge is removed and substrate is replaced, while maintaining the above engineering considerations. Any habitat value should reflect the potential uptake of toxic metals to birds, riparian mammals, and amphibians while enhancing the aesthetic quality of the project.

Many aerobic wetland systems have enjoyed long-term success and cost effectiveness. However, there have also been many failures, which have been very damaging to their perceived effectiveness. In general, systems that were not effective or failed were undersized, improperly designed, or both. The key, as with all water treatment systems, is to understand the limitations of each unit's operation, to have reasonable expectations, and to use conservative sizing criteria to attain specific water quality goals. Even undersized passive systems can be useful, discharging water with significantly lower concentrations of metal contaminants than were present in the inflow drainage. These improvements in water quality have significantly decreased the costs of subsequent water treatment at active sites, and deleterious impacts that discharges from abandoned sites have on receiving streams and lakes.

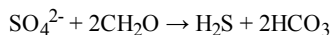
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7.5.2.2 Anaerobic Wetlands and Biochemical Reactors

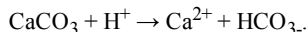
Anaerobic systems primarily rely on chemical and microbial reduction reactions to precipitate metals and neutralize acidity. The water infiltrates through thick, permeable organic material that becomes anaerobic due to high biological oxygen demand. Several other treatment mechanisms function beyond those in aerobic wetlands, including metal exchange reactions, formation and precipitation of metal sulphides, microbially-generated alkalinity due to reduction reactions, and continuous formation of carbonate alkalinity due to limestone dissolution under anoxic conditions. Since anaerobic wetlands produce alkalinity, their use can be extended to poor quality, net acidic, low pH, high Fe, and high dissolved oxygen (>2 mg/L) ARD. Microbial mechanisms of alkalinity production are of critical importance to long-term ARD treatment. When wetlands receive high acid loads (>300 mg/L), the pH-sensitive microbial activities are eventually overwhelmed. Therefore, like their aerobic counterparts, anaerobic wetlands are most successful when used to treat small ARD flows and/or ARD that has moderate water quality.

The ARD treatment mechanisms for anaerobic biochemical reactors (BCRs) (also referred to as compost reactors) are based on alkalinity addition using the following two mechanisms:

- Sulphate reduction, which converts SO_4^{2-} into H_2S in an organic rich environment devoid of oxygen, releases alkalinity as a by-product as follows:



- Limestone and dolomitic material react to neutralize acidity as follows:



Carbonate material also suppresses fermentation bacteria, which are required in the bacterial consortium, but are not desirable in quantity, since fermentation by-products can lower the pH.

The key features of an anaerobic biochemical reactor are:

- A substrate bed containing a varied blend of natural material (e.g., wood chips, crushed limestone, plant residue, grass cuttings, hay, straw, manure, and compost)
- A surface pond (at least 150 mm deep), which floods the substrate bed and limits oxygen ingress into the BCR
- Mine water flow distribution and collection system to promote a plug flow pattern (typically configured vertically) with limited risk of short circuiting or dead zones
- Flow and level control devices to control the water level and to prevent substrate from being exposed to the atmosphere
- Higher plant life may be present to assist with organic material supplementation, as a wildlife habitat and for aesthetic appearance. However, vegetation may need to be suppressed in BCRs with a relatively thin (< 750 mm) substrate layer because the oxygen infusion from the plant activity can impact the establishment of geochemical reducing conditions.

BCRs constructed in the 1990s were typically horizontal plug-flow cells that resulted in a significant amount of mine water flow across the cell surface. These were often referred to as compost wetlands (Hedin et al., 1994). The current common practice is to use a vertical flow configuration with untreated water introduced at the top of the cell and treated water collected from the bottom.

The mechanisms of metals removal vary depending on the specific metal, but mechanisms of metals removal are a combination of the following:

- Sulphide precipitation
- Oxidation/hydrolysis (on the BCR surface if iron is present)
- Carbonate precipitation
- Absorption onto organic matter

A key advantage of BCRs is that the organic matter is typically found locally, as is the consortium of bacteria that populate the substrate. Common animal manure (browsing animals like cows, sheep, or goats are preferred) provides the bacterial inoculum for these units.

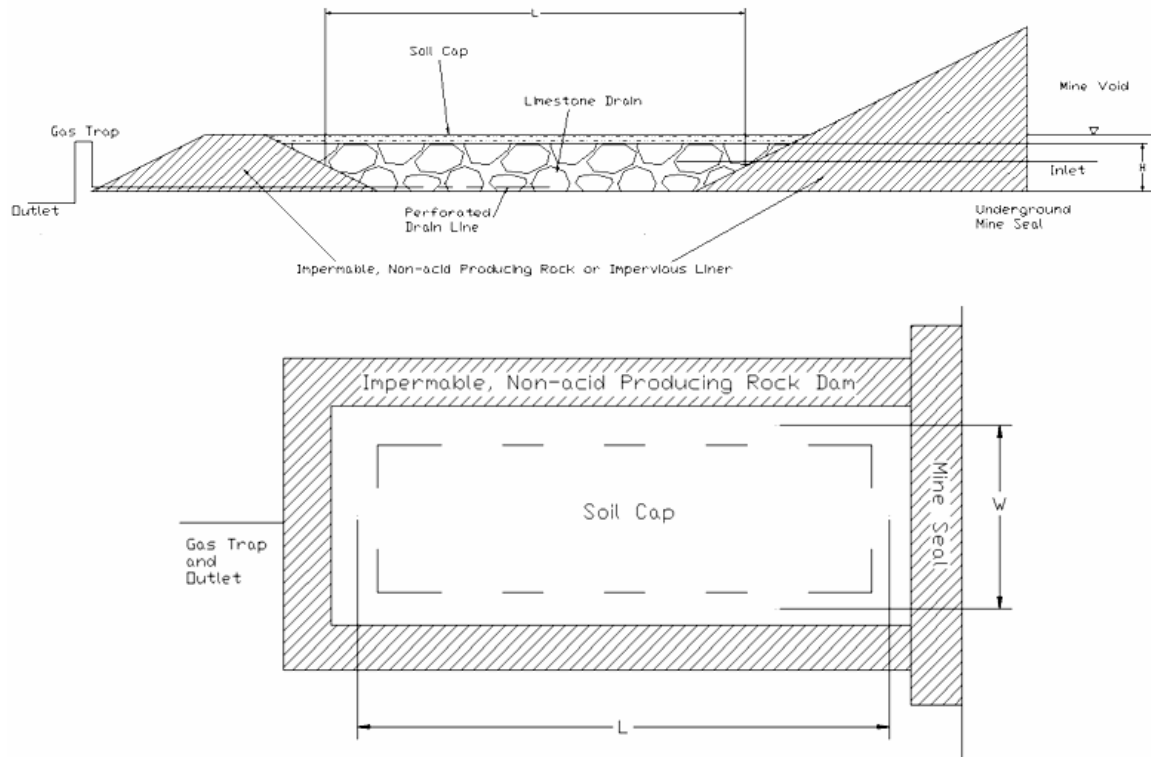
BCRs are typically followed by aerobic cells. Systems are typically comprised of two BCRs to facilitate long-term maintenance (all flow is temporarily directed to one BCR, while the other is being retrofitted) feeding into a single multiple-compartment aerobic wetland.

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7.5.2.3 Anoxic Limestone Drains

Anoxic limestone drains (ALDs) are buried cells or trenches of limestone into which anoxic water is introduced (Figure 7-16). The ALD must be sealed so that the inputs of atmospheric oxygen are minimized, and the accumulation of CO_2 within the ALD is maximized. This is usually accomplished by burying the ALD under 1 to 3 m of clay. Plastic is sometimes placed between the limestone and clay as an additional gas barrier. In some cases, the ALD has been completely wrapped in plastic before burial (Skousen and Faulkner, 1992). This can also help keep clay and dirt from entering the pore volume from the bottom and sides of the excavation.

Figure 7-16: Anoxic Limestone Drain Design



The limestone dissolves in the acid water, raises pH, and adds alkalinity. Under anoxic conditions, the limestone does not coat or armour with Fe hydroxides because Fe^{2+} does not precipitate as $\text{Fe}(\text{OH})_2$ at an acid or circum-neutral pH. In addition to little or no dissolved oxygen and Fe^{3+} , aluminum concentrations must also be low: less than 2 mg/L. An ALD in western Pennsylvania that received 21 mg/L of aluminum completely clogged in eight months.

Limestone with higher CaCO_3 content (> 80%) dissolves faster than limestone with a higher MgCO_3 or $\text{CaMg}(\text{CO}_3)_2$ content ($\approx 50\% \text{CaCO}_3$) (Watzlaf and Hedin, 1993). The limestone used in most successful ALDs contains 80 to 95% CaCO_3 . Most effective systems have used 5- to 20-cm-sized limestone. Some systems constructed with fine and small gravel limestone have failed, apparently because of plugging problems.

The ALD should be designed to inundate the limestone with water at all times. Clay dikes within the ALD or riser pipes at the outflow of the ALD will help ensure inundation. Also, the ALD discharge should be equipped with a plumbing trap to prevent air from entering the system. Finally, a pond must be constructed down-gradient to capture all of the iron that will precipitate once the neutralized water contacts the atmosphere. Typically, this pond is followed by additional ponds or wetlands to further enhance water quality. The dimensions of ALDs vary considerably. Narrower ALDs have the advantage of minimizing short-circuiting, but present a small cross-section perpendicular to the flow and thus may be more prone to clogging. Wider ALDs may be less likely to suffer significant permeability reductions (clogging) but may allow short-circuiting to occur. Site conditions will often dictate the dimensions of the ALD.

Faulkner and Skousen (1994) reported both successes and failures among 11 ALDs treating mine water in West Virginia. In all cases, water pH was raised after ALD treatment but three of the sites had pH values <5.0, indicating that the ALD was not fully functioning. When working correctly, the pH values of water in ALDs should be at least 6.0. Water acidity in these drains decreased 50 to 80%, but Fe and Al concentrations in the outflow, unfortunately, also decreased. Ferric iron and Al precipitate as hydroxides at this pH; reductions in dissolved Fe and Al indicate that some coating or clogging of limestone likely occurred.

Thus, longevity of treatment is a major concern for ALDs, especially in terms of water flow through the limestone. Unless there is no Fe^{3+} , dissolved oxygen, or Al present, eventual clogging of the limestone pore spaces with precipitated Al and Fe and/or gypsum is predicted at many sites (Nairn et al., 1991). Selection of the appropriate water and environmental conditions is critical for long-term alkalinity generation in an ALD. Like wetlands, ALDs may be a solution for ARD treatment for specific water conditions or for a finite period, after which the system must be replenished or replaced.

Tracer studies indicated that while ALDs approximate plug-flow systems, some short-circuiting occurs, and dead areas do exist. Calculated retention times, using 49% porosity, were in fairly good agreement with the median retention times of the tracer tests (Watzlaf et al., 2004). Water quality data determine the applicability of an ALD and flow data provide the basis for sizing an effective ALD for the desired design life. Approximately 15 hours of contact time between mine water and limestone in an ALD is necessary to achieve a maximum concentration of alkalinity. In order to achieve 15 hours of contact time within an ALD, 2,800 kg of limestone is required for each L/min of mine water flow. For example, an ALD that discharges water with 300 mg/L of alkalinity (the maximum sustained concentration thus far observed in an ALD effluent), dissolves 1,750 kg of limestone (90% calcium carbonate) in ten years, per each L/min of mine water flow. Therefore, a limestone bed should contain 6,200 kg of limestone for each L/min of flow (equivalent to 26 tons of limestone for each gallon per minute of flow). This assumes that the ALD is constructed with 90% CaCO_3 limestone rock and has a porosity of 49%. The calculation also assumes that the original CMD does not contain Fe^{3+} or Al. The presence of these ions could result in faster rates of limestone dissolution due to the generation of acidity during hydrolysis. More importantly, they have the potential to limit limestone dissolution and cause a significant reduction in permeability that could very well lead to failure (as previously discussed). For a more detailed discussion of limestone dissolution rates, see Cravotta and Watzlaf (2002).

To summarize, the success of ALDs depends on the following:

Iron must be in the reduced ferrous (Fe II) form because ferric iron (Fe III) will armour the limestone material (if not, use a RAPS, as described in

Section 7.5.2.3)

No free oxygen (< 1 mg/L) must be present; otherwise iron (Fe III) precipitation will take place (see RAPS in Section 7.5.2.4, if water is oxygenated)

Low mine water aluminum concentration (< 2 mg/L) because any aluminum hydroxide precipitates will clog the limestone bed

A vent for excess CO₂ formed in the ALD

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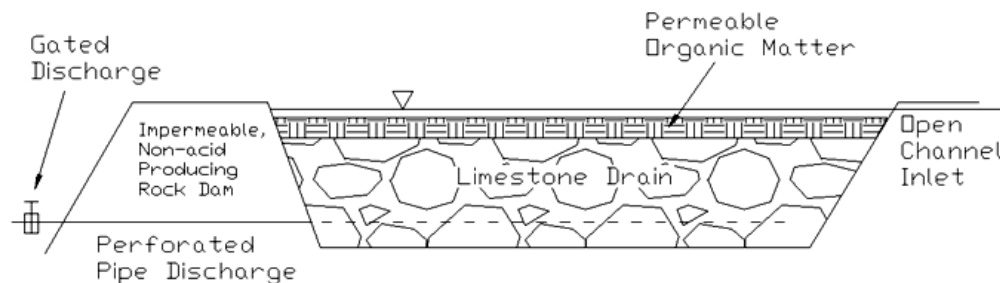
7.5.2.4 Reducing and Alkalinity Producing System Wetlands

If the water contains dissolved oxygen or ferric iron, a reducing and alkalinity producing system (RAPS) will function better than an anoxic limestone drain. RAPS are similar in construction to an anaerobic BCR, but the function of a RAPS is to reduce ferric iron to ferrous in a thin organic layer (as opposed to a much thicker substrate layer in the BCR) and then neutralize the acidity in a limestone layer installed beneath the organic layer. Sulphate reduction also takes place, which generates alkalinity, and can precipitate some metals as sulphides. However, alkaline addition in RAPS is dominated by the limestone dissolution pathway. The acid neutralization potential afforded by a RAPS ranges from 35 to over 400 mg/L CaCO₃. Sulphate reduction contributes an average of 28% (with a range of 5 to 51%) of the total alkalinity produced. The rate of alkaline addition for a single RAPS unit is about 40 to 60 gd⁻¹. The rate of alkaline addition for a second RAPS unit in a series is about 1/2 to 1/3 of the rate of the first unit.

This type of system was first implemented at Galax, Virginia, in the late 1980s to treat highly acidic, high-iron water emerging from an abandoned pyrite mine (Hendricks 1991). In 1991, a second system of this type was constructed to treat water being discharged by a coal processing waste landfill near Norton Virginia (Duddleston et al., 1992). The term "successive alkalinity producing system (SAPS)", indicating that more than one of these units could be used in series to treat very highly acidic water, was applied to these systems by Kepler and McCleary (1994), who demonstrated a successful application at the Howe Bridge site in NW Pennsylvania. The Kepler and McCleary application received widespread notice, and use of these systems expanded rapidly thereafter. Similar systems have also been referred to as vertical flow systems, vertical flow ponds, or vertical flow wetlands. Chemically, biologically, and physically, these systems behave similarly, and are all referred to here as RAPS since most applications involve just a single system followed by an oxidation pond to precipitate and settle iron from the alkalinity-buffered RAPS effluent.

A typical design involves a sedimentation pond or aerobic wetland to precipitate any suspended ferric hydroxide that may be present. This is followed by the RAPS, which is constructed by placing a layer of limestone (0.6 to 1.2 m thick) on the bottom of an excavated area. A network of perforated pipes is placed in the lower portion of this limestone layer. Organic material (0.15 to 0.6 m thick), which typically has been composted, is placed above the limestone, and serves as the nutrient source for the iron- and sulphate-reducing bacteria. The composted organic material lies beneath 1-3 m of water (Figure 7-17; the water pressure helps force the water through the organic layer).

Figure 7-17: Profile view of a reducing and alkalinity producing system (RAPS) (not to scale)



RAPS are now more common than ALDs for treatment of CMD because they are appropriate for water that contains dissolved oxygen or ferric iron, which can armour the limestone in an ALD. It is thought that RAPS may also be more resistant to plugging by aluminum than ALDs because of their larger cross sectional area and higher available head pressures (Watzlaf and Hyman, 1995). The Howe Bridge RAPS treated water for 11 years before being replaced. After 11 years, it was still able to pass 50% of the influent water through the compost and limestone layers. However, this system received less than 0.2 mg/L of aluminum. It appeared that the progressive reduction in permeability was due to the lack of a preliminary sedimentation pond; iron hydroxides precipitated on top of the compost layer, with an eventual accumulation of more than 15 cm of iron sludge on top of the compost. Reduced permeability can also result from storm-mobilized silt and other solids, as well as precipitation of metal sulphides within the organic layer. Thus, continued monitoring of the actual performance of these systems is warranted.

Kepler and McCleary (1997) described a flushing mechanism that they reported allowed RAPS to resist clogging by aluminum. However, field experiments conducted by Watzlaf et al. (2003, 2004) indicated that, although it appears that significant solids are being flushed out, the actual amount is only a minor component of what apparently precipitated in the system, based on water quality records. So, if aluminum is present at significant concentrations in the mine water, this alkalinity-adjusting method should be avoided because of potential plugging.

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7.5.2.5 Open Limestone Drain

Open limestone drains (OLDs) are designed to introduce alkalinity into the dissolution of exposed limestone in the bottom and sides of a limestone drain. Past assumptions held that limestone armoured or coated with Fe or Al hydroxides ceased to dissolve. Ziemkiewicz et al. (1994, 1997) reported that armoured limestone was still somewhat effective (50 to 90%, compared to unarmoured limestone), and that seven OLDs in the field reduced acidity in ARD by 4 to 62% compared to a 2% acid reduction in a sandstone channel. They suggested that OLDs would be useful in abandoned mine reclamation projects where one-time installation costs can

be incurred and regular maintenance is not possible.

Long channels of limestone can be used to convey ARD to a stream or other discharge point. Based on flows and acidity concentrations, cross sections of stream channels (widths and heights) can be designed with calculated amounts of limestone (which will become armoured) to treat the water. However, the design and operation of the limestone drain require special attention to accommodate the inevitable armoring and coating of the limestone. The following features in open limestone drain are recommended:

- Steep drain slopes of > 20%
- High flow velocities to scour settled solids and clean precipitates from the limestone surfaces
- Ability to periodically flush the OLD and clear accumulated precipitates and solids

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7.5.2.6 Passive Sulphate Removal

A special category of passive treatment technology has been specifically developed to achieve high rates of sulphate reduction and ultimately sulphate removal as elemental sulphur. While anaerobic wetlands do incorporate a degree of sulphate reduction, rates are low and these wetlands are without a dedicated oxidative process to remove the sulphides as elemental sulphur. An integrated passive mine water treatment process has been developed in South Africa (Pulles et al., 2004) using integrated and managed passive treatment (IMPI). The IMPI technology has not been applied to many full-scale and permanent treatment sites. Passive sulphate removal uses the same fundamental treatment mechanisms at work in an anaerobic wetland, but with some of the following novel features:

- The Degrading Packed Bed reactor is filled with a specific sequence of selected organic materials, designed to hydrolyze ligno-cellulosic materials. The objective is to sustainably produce volatile fatty acids (VFA) to drive the sulphate reduction process.
- The sulphide oxidizing reactors (primary and secondary) are intended to partially oxidize the H₂S to sulphur, with limited impacts on the VFA concentrations.
- The sulphate reduction reactor relies on the upstream generation of adequate and suitable readily biodegradable compounds, such as VFAs to support the sulphate reducing bacteria.

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7.5.2.7 Alkaline Leach Beds

Alkaline leach beds are ponds or cells filled with limestone or steel slag. Like OLDs, they have occasionally been used to improve water quality at abandoned mine sites. The alkalinity is added up-gradient of significant concentrations of dissolved metals. Ideally, slightly acidic water with no metals is introduced into limestone-filled ponds. The limestone dissolution adds 50 to 75 mg/L of alkalinity as CaCO₃ to the water. The alkalinity buffers the stream and mitigates the effects of ARD entering downstream. At several sites where limestone-filled alkaline leach beds have been installed, fisheries have been re-established.

In situations where large metal and acid loads enter downstream, the upstream water must be charged with greater levels of alkalinity. Steel slag, a by-product and waste from the making of steel, contains high levels of alkalinity that are released into water. Alkaline leach beds can be filled with steel slag, which can generate much higher alkalinities in water (as much as 2,000 mg/L as CaCO₃). Sites where these high alkalinities are generated must be carefully selected, because water that is too alkaline can be toxic to aquatic life.

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7.5.2.8 Manganese Oxidation Beds

Manganese oxidation beds (MOBs) appear similar to alkaline leach beds but they are positioned as the final step in a successful passive treatment system of CMD. MOBs support the growth of a bacterial/algal consortium. The initial precipitation of MnO₂ or similar compounds is slow, but is apparently aided by the bacterial activity. Because MOBs are intended to facilitate manganese oxidation, the limestone cannot be completely inundated; the general rule is "one should be able to easily walk across a MOB without getting your feet wet". Both research and experience have indicated that the bacterial/algal organisms are naturally occurring and will typically colonize the bed within six to eight weeks (Brant and Ziemkiewicz, 1997, Rose et al., 2003). Once the bacteria oxidize the manganese and induce manganese oxide precipitation, the mineral surface catalyzes additional manganese oxidation (auto-catalysis). The algae employ the MnO₂ formed to provide hold-fasts to rocks in the flowing water and appear to facilitate manganese removal.

MOBs only function as a polishing step in a passive treatment system because they are only effective after virtually all of the iron has been removed, since dissolved Fe²⁺ chemically reduces manganese, causing it to re-dissolve. Also, while MOBs allow manganese to be inexpensively removed at circum-neutral pH, manganese is sometimes only regulated as a surrogate for other more toxic metals, as stated in the CMD section of Chapter 2. Where that is the case, the presence of such metals may argue against the emplacement of a MOB unless the removal of those metals has also been addressed.

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7.5.2.9 Design of Passive Treatment System Components

Gusek (2008) provides a detailed overview of the testing required to design a passive treatment system, which is summarized below. If the chemistry of the acidic drainage is complex or unique, the initial phase of the passive treatment evaluation might occur in the laboratory. As with any treatment process design, the composition of the water to be treated, the nominal flow rates and seasonal variations, and the target effluent levels must be clearly defined.

Initial Feasibility Testing: Typically, locally-available and plentiful candidate substrate materials for BCR systems are evaluated in the laboratory, involving testing,

utilizing about 30 to 60 grams of different substrate materials in culture bottles immersed in drainage samples. The tests take about six to eight weeks. Aerobic testing is typically conducted simultaneously by monitoring effluent behaviour over time under aerobic conditions (algae inoculum) without substrate. A typical algae inoculum may include pond scum or algae growths from natural wetland sites near the project. Indicative measurements during proof-of-principle testing include pH, oxidation reduction potential (ORP), conductivity, substrate/water color, and odour (Gusek, 2008). These studies are static rather than flow-through experiments and are typically developed to test the suitability of the candidate substrate materials or inoculum in a passive treatment component and determine whether removal of a contaminant by microbial processes in a wetland with a known substrate composition is possible.

Bench Scale Testing: To conduct effective bench-scale tests, approximately 100 kg of substrate are operated in the field for at least three months utilizing a typical range of dissolved metals concentration in the influent. This approach begins to simulate the typical kinetic chemical reactions that might occur at a larger scale. Site-specific loading factors and substrate hydrology/permeability characteristics are determined during bench-scale testing.

Pilot-Scale Testing: Successful bench-scale testing supports the construction of pilot-scale systems utilizing tonnes of substrate. These systems are typically operated for at least a year before full-scale system design is finalized. If possible, pilot system cells are sized to be integrated into the overall, operational passive system design.

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7.5.2.10 Passive Treatment System Performance

Generally, operational problems with passive treatment systems can be attributed to inadequate design, unrealistic expectations, pests, inadequate construction methods, and/or unanticipated perturbations (e.g., extreme storm events, long droughts). If properly designed and constructed, most passive treatment systems function very well with a minimum amount of attention and money. However, the specific performance and useful life of passive treatment systems are difficult to predict with a high level of confidence. The treatment kinetics and efficiency of such systems are influenced by site-specific environmental conditions, flow conditions and patterns, complex natural organic material, water chemistry, and seasonal variability. It is therefore important to pilot test such technologies before full-scale implementation and to use conservative design criteria and performance estimates.

Design and operation of passive treatment systems must take into account seasonal variations and specifically cold climate winter conditions. All biochemical and microbial reaction rates decrease as temperatures drop. Freezing conditions will impact passive treatment system performance, and can cause the system to fail. Care must be taken in applying generic design criteria to such cold winter operating conditions. Precautions can be taken in the case of some passive treatment facilities (such as anoxic limestone drains and RAPS) to insulate the treatment unit against the extremes of winter temperatures. However, the mine drainage temperature may still decrease during winter and impact the treatment efficiency. Pilot testing over a full year or more should provide data on efficiency changes in response to depressed mine water temperatures, if they can be anticipated to occur.

Limited information is available from full-scale treatment processes operated for a sustained period of time on the removal efficiencies for some contaminants. Younger et al. (2002) compiled a summary of postulated passive treatment removal mechanisms, which has been enhanced and modified based on the prevailing wisdom and experience regarding these systems in Table 7-8.

Table 7-8: Postulated Removal Mechanisms of Metals and Mining-related Pollutants in Passive Treatment Systems

Parameters	Postulated Removal Mechanisms	
	Aerobic Wetlands	Anaerobic Wetlands
Arsenic	Oxidation to form AsO_4^{3-} , adsorption to ferric oxides	Reduction to As^{3+} , precipitated as a number of sulphides
Cadmium	-	Precipitation as a sulphide
Chromium	-	Reduction of Cr^{6+} to Cr^{3+} , precipitated as hydroxide
Copper	Oxidation in alkaline environment, precipitation as carbonate	Reduction and precipitation as sulphide
Cyanide	Photolytic conversion, bacterial oxidation to NH_3 and N_2	Reduction and decomposition to NH_3 and CO_2
Lead	Oxidation in alkaline environment, precipitation as carbonate	Precipitation as a sulphide
Nickel	-	Precipitation as a sulphide
Zinc	Precipitation as a carbonate	Precipitation as a sulphide

The removal of these contaminants takes place simultaneously with the mainstream processes of removal of acidity, iron, sulphate, and aluminum, if present. The available information on the removal rates of the non-ferrous metals and other mine water parameters is growing, but site-specific verification is highly recommended. System designers must account for the potential for portions of a passive system to lose effectiveness, to determine where uncontrolled release may occur, and must allow for the long-term depletion of neutralizing material, such as limestone chips.

Probably the most common maintenance problem is stability in the dike and spillway. Reworking slopes, rebuilding spillways, and increasing freeboard can all be avoided by proper design and construction using existing guidelines for such construction.

Pests can plague wetlands with operational problems. Rodents such as muskrats can burrow into dikes, causing leakage and potentially catastrophic failure problems,

and can also uproot significant amounts of cattails and other aquatic vegetation. Muskrats can be discouraged by lining dikes and slopes with chain link fence or riprap to prevent burrowing (Brodie, 1990). Beaver dams can cause water level disruptions and can seriously damage vegetation. They are very difficult to control once established. Small diameter pipes traversing wide spillways (three-log structure) and trapping have had limited success in beaver control. Large pipes with 90-degree elbows on the upstream end have been used as discharge structures in beaver-prone areas (Brodie, 1991). Otherwise, shallow ponds with dikes and shallow slopes toward wide, rip-rapped spillways may be the best design to deter beaver populations. Insects, such as the armyworm, with their appetite for *Typha*, have devastated monocultural wetlands (Hedin et al., 1994a). The use of a variety of plants in a system will minimize such problems. Mosquitoes can breed in wetlands where mine water is alkaline. In southern Appalachia, mosquito fish (*Gambusia affinis*) have been introduced into alkaline-water wetlands to control mosquito populations (Brodie, 1990).

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7.5.2.11 Concluding Comments on Passive Treatment

Characterization of influent water quality and quantity, including seasonal variation, is important prior to the selection and development of a passive treatment system (Hyman and Watzlaf, 1995). The presence or absence of periodic events, such as spring flushes of deposited metal salts from within the mine area, may influence the selection and sizing of passive systems.

Aerobic ponds and wetlands can be very effective for the removal of iron from net alkaline mine water, especially CMD. It appears that the original estimate of Hedin et al. (1994a) of 10 to 20 $\text{gd}^{-1} \text{m}^{-2}$ remains a convenient pre-construction rule-of-thumb for estimating pond and wetlands sizes. Recent studies have provided insight into the factors that control the overall processes, and these approaches may be used to fine-tune sizing criteria. As stated earlier, aeration can be used to sparge CO_2 and increase pH, which can significantly increase iron oxidation rates, thereby reducing the size of aerobic ponds and wetlands needed for iron removal.

ALDs can effectively treat net acidic mine water with a pH below 5.0. At this pH, ferric iron and aluminum concentrations will be very low. Intercepted ground water is typically low in dissolved oxygen, and often contains partial pressures of CO_2 higher than atmospheric levels, which allows for development of alkalinity concentrations greater than 100 mg/L as CaCO_3 . Near maximum levels of alkalinity (usually between 150 and 300 mg/L) can be achieved with 15 hours or more of contact time. ALDs are tolerant of both ferrous iron and manganese, because they remain soluble within the ALD. However, the presence of ferric iron, and particularly aluminum, can reduce permeability of the ALD by precipitation of these metals within the voids in the limestone. In the absence of ferric iron and aluminum, ALDs have continued to perform well with no obvious seasonal variation or long-term reduction in effectiveness.

At mine sites where the appropriate water quality criteria were met and the ALD was sized properly, effective treatment of mine drainage occurred, provided that the ALD was followed by ponds and/or wetlands for iron oxidation, precipitation, and settling. At these sites, it is projected that the ALD will be effective for the designed lifetime of 25 to 30 years and, in some cases, well beyond. ALDs offer an effective means of introducing alkalinity into net acidic waters that contain neither ferric iron nor aluminum. The presence of either of these ions will reduce permeability of the ALD by precipitation, which will cause premature failure by clogging. In the absence of these ions, ALDs have continued to perform well with no obvious seasonal variation or long-term degradation. Near maximum levels of alkalinity (usually between 150 and 300 mg/L) can be achieved with 15 hours or more of contact time. ALDs are tolerant of both ferrous iron and manganese. ALDs must be viewed as a unit operation, not a standalone remediation technique, and must be followed by a pond and wetland for iron oxidation, precipitation, and settling.

Alkaline addition in RAPS is dominated by the limestone dissolution pathway. The rate of alkaline addition for a single RAPS unit is about 40 to 60 $\text{gd}^{-1} \text{m}^{-2}$. Rates for the second RAPS in a series fall off to about 1/2 to 1/3 of the rate of the first system. Much of the variability in performance can be attributed to influent water quality and detention time. As with ALDs, RAPS should be viewed as unit operations, not stand-alone technologies. They must be preceded by a pond/wetland to precipitate iron and other settleable solids. As with ALDs, RAPS must also be followed by a pond and wetland for iron oxidation, precipitation, and settling.

Finally, care should be taken to obtain sufficient water quality data, including seasonal variation, before designing and developing a passive treatment system. Site and funding constraints may limit the applicability of passive techniques for some mine drainages. However, for those drainages with appropriate water quality and land availability, passive treatment systems continue to perform very well.

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7.5.3 In situ Treatment Technologies

In situ treatment of mine drainage can be undertaken in many different ways and configurations. This section is limited to a brief discussion that includes the following:

- Spreading of alkaline material across mining impacted land and mine waste
- In pit water (pit lake) treatment
- Organic covers of mine land and mining waste
- Permeable reactive barriers (i.e., organic-rich material, zero-valent iron)

In situ treatment of acidic mine water by injection of alkaline lime slurry to disturbed mine land, spoils, and mining waste has met with mixed success for mine drainage. The challenges to practical mine-scale applications include the following:

- Flow and transport characteristics of the mine waste material, described as pseudokarstic aquifer due to the presence of interconnected preferential flow paths
- Introducing the lime slurry (or any other alkaline solution) in a manner that will ensure distribution and effective contact with acid producing zones or water bodies
- The scale of such operations and the preparation of infiltration beds or trenches, which do not blind or suffer from ponding

Full-scale trials have been conducted in West Virginia surface coal mines (Donovan et al., 2000) with some success. Pit lake treatment typically involves the spreading and dispersion of an alkali material across the accumulated water surface. The challenge is to effectively bring the alkali material into contact with the large water body. Available approaches are as follows:

Approach 1 - Spreading of the alkali material in a powder or slurry form across the full aerial extent of the pit lake. This relies on the even spreading of the alkali material and sufficient mixing and contact time between the alkali material and the pit lake. Unreacted alkali material will accumulate on the pit floor, together with associated neutralization reaction products such as metal precipitates.

Approach 2 - Abstracting the pit water and pumping/flowing the water across or through an alkali mix device for blending and dissolution of alkali material. The pit water and alkali blend stream is then returned to the pit for completion of the neutralization reactions, precipitation of metals, and dispersion of the alkali material.

Approach 3 – Adding alkali material in the early stages of pit flooding as water is entering the pit or workings.

The challenges to in situ pit water treatment include the following:

- Effective contact between alkali material and pit water
- Efficient use of available alkali material
- Long-term dissolution of precipitated metals from the sludge layers
- Poor control of the pH and redox conditions in all parts of the pit lake to achieve the target treatment objectives

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7.6 Treatment Residues and Wastes

All mine drainage treatment technologies produce some residues (e.g., sludge, brines, and spent media) or emissions (e.g., gasses). These residues and emissions contain the elements and compounds removed from the mine drainage and the additives and supplements dosed in the treatment process. No consideration of mine drainage treatment technologies is complete without an understanding of these residues and emissions as it relates to the following:

- Relative production in terms of volumes and masses
- Typical characteristics in terms of chemical composition (e.g., hydroxide, sulphide, and NP) and physical properties (i.e., consistency, volatility, and dewater ability)
- Hazardous classification and rating
- Potential environmental impacts
- Disposal options

The treatment residues can be broadly classified into the following two categories:

- Sludge, which is a slurry or dewatered cake containing precipitates of diverse composition
- Brines, which contains soluble salts in high concentrations

7.6.1 Sludge Management

Sludge management is an escalating concern as the inventory of sludge continues to increase and the stability of the sludge under various disposal conditions is poorly understood. As such, the management and disposal of these mining wastes requires careful consideration and planning.

7.6.1.1 General Considerations

To design the most appropriate sludge management strategy for a site, several factors need to be considered. The principal considerations are the mass of sludge produced, whether the mine is operating or closed, dewatering ability of the sludge, sludge density (moisture content), sludge volume, chemical and physical stability, sludge composition, disposal location availability, and economics (Zinck, 2006). The ability of a sludge to dewater may limit the options available. Sludges that can dewater without mechanical assistance will not only reduce the area required for disposal, but also make it more attractive for reuse options. The ability of sludge to dewater depends on its particle size, morphology, and surface charge. As a particle deviates from a spherical shape, the surface area per unit volume increases, resulting in reduced settleability and decreased dewatering rate. These characteristics are linked directly to the water treatment process that generates the sludge and to the raw water chemistry (Zinck, 2005).

7.6.1.2 Sludge Disposal

Various options available for sludge disposal are reviewed below.

Pond disposal

Sludge management involves three principal steps, namely solid-liquid separation, sludge dewatering, and disposal. Many sites utilize settling ponds as an efficient sludge management option. The sludge is pumped to a settling pond where solid-liquid separation, dewatering and, in many cases, disposal occur simultaneously. Issues associated with pond disposal are minimal. Wind resuspension and dusting present problems at some sites, particularly in arid or northern regions. Due to the large requirement for space, land use can be a challenge for some sites. Due to the thixotropic nature of sludges (viscosity decreases as shear strength increases), pond failure could present some concerns, although generally not to the same extent as with tailings impoundments. In a pond environment, either with or without a water cover, the degree of metal leaching is expected to be minimal, as the excess alkalinity available in the sludge is enough to sustain a moderate pH for decades, even centuries (Zinck et al., 1997). Sludge disposal in a pond environment can be either subaerial or subaqueous. In a subaerial environment, the sludge is exposed to weathering conditions. Sludge cracking due to moisture loss at the surface is prevalent, causing an increase in surface water infiltration. Under these conditions, sludge dewatering occurs at the surface while the majority of the sludge at depth is still very moist. The desiccated surface may be reclaimed (Zinck, 2006).

Codisposal with Tailings

The practice of co-mixing tailings with treatment sludge for disposal involves injecting the treatment sludge into the tailings slurry prior to discharge to the impoundment. Typically, the sludge to tailings ratio is less than 1:20. Here the sludge serves to fill void spaces within the tailings, in theory reducing the potential for water or air

infiltration and the hydraulic conductivity of the mixture. This method of disposal could be an effective option provided that the tailings are either non-acid generating or that tailings oxidation is prevented. However, if the tailings undergo oxidation and commence acid generation, the likelihood for sludge dissolution and metal mobilization is high (Zinck, 2006).

Sludge as a Cover over Tailings

The application of wet and dry covers to prevent acidic drainage is widely adopted. Wet covers provide a barrier that minimizes oxygen contact with potentially acid generating material and, except for minor oxygen dissolved in the water, precludes contact with atmospheric oxygen completely. Some of the issues related to the application of a sludge cover on tailings are cracking and preferential channelling. Therefore, sludge needs to be disposed in a manner by which the particles will not segregate, such that the sludge and the underlying tailings remain saturated (Zinck, 2006).

Sludge Disposal with Waste Rock

Disposing sludge with waste rock has several of the same potential benefits as disposal with tailings, including utilization of excess alkalinity to offset acid generation and filling of void spaces. This practice of disposing treatment sludge in waste rock piles is being adopted at some sites. While results (Coleman et al., 1997) show that sludge is not effective as a capping material, this method was found to be a low-cost final disposal option because the sludge filled pore spaces and voids within the waste rock pile.

Disposal in Underground Mine Workings

Disposal of treatment sludge into underground mine workings has several benefits that make it an attractive sludge management option. The deposition of sludge into underground mines reduces the footprint required for disposal sites (landfills and impoundments), eliminates the potential for surface water pollution, reduces the potential for subsidence, and improves the aesthetics of the local area. Also, in acidic mine workings, the underground disposal could have the additional benefit of reducing the acidity of the mine water. This practice involves pumping or trucking sludge to boreholes, which are drilled into underground inactive mines. Some of the factors that need to be considered in this disposal option include:

- Site availability and access
- Mine capacity, void space, configuration
- Sludge properties (e.g. viscosity)

This method is very attractive from an economic and environmental standpoint. However, like most disposal options presented, this is clearly site specific. Sludge with high iron content can most probably be disposed of this way economically. Disposal of sludge with high Cd, Zn, or Ni content in this manner may or may not be economic or environmentally acceptable depending on the contact effectiveness and ratio between the sludge and acidic mine water, the alkalinity of the sludge, and the acidity of the mine drainage (Aubé, 2004 and Aubé et al., 2005).

Disposal in Pit Lakes

Disposal in an abandoned open pit is typically one of the most economical solutions for sludge storage, if a pit is within a reasonable pumping distance from the treatment plant. Many companies frequently take advantage of open pits available on site as an appropriate short or long-term sludge disposal option. Some excellent work on this option has been described by McNee et al. (2003) and McNee (2004).

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7.6.2 Brine Management

Brine disposal is much more challenging, and the disposal options include the following:

- Incorporation into a mine waste or tailing stream
- Irrigation and potential cultivation of salt resistant plants
- Solar evaporation ponds, possibly with some wind-assisted features
- Discharge and dilution in a sanitary sewer
- Mechanical evaporation and crystallization
- Beneficial use in the cultivation of halophilic (“salt loving”) algal species of commercial value

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7.7 Recovery of Useful Products

A paradigm shift has taken place in the handling and management of treatment residues, such as sludges and brines. The recovery of useful and saleable products is now researched and actively pursued. The recovery of useful products from the treatment process waste streams may include the following:

- Metals recovery
- Supplements for mine land rehabilitation and revegetation, such as $\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$
- Alkali recovery, such as CaCO_3
- Building and construction related materials, such as gypsum
- Beneficial use of brine in the cultivation of halophilic organisms, such as algae containing high β -carotenes and other nutritional supplements
- Recovery of saleable products, such as sulphur and magnesium salts
- Agricultural use (e.g., fertilizer)
- Supplement in cement manufacturing
- Gravel from sludge

Metal adsorbents in used industrial wastewater treatment
Pigment (ferrihydrite) (Hedin, 1988, 2002)

For a detailed discussion on reuse options for ARD treatment sludge refer to Zinck (2005). Research and development work in this area are ongoing. The incentives driving the recovery of byproducts include the following:

- Reduction of waste sludge and brine products, which require perpetual handling and disposal with associated long-term environmental liabilities
- Generation of a revenue stream to partly or fully offset the ongoing treatment cost
- Contribution to the long-term sustainability of mine water treatment projects

The key aspects of successful byproducts recovery in the treatment of mine drainage are as follows:

- The target byproducts must be selectively removed by minimizing the co-precipitation of compounds that would degrade the quality of by-products.
- By-products recovery, as a project objective, will have an impact on the mainstream treatment process in terms of unit treatment, process selection, and sequence of treatment processes.
- Chemicals (reagents) dosing to the mainstream treatment process must take into account the impact on the potential for and composition of by-products.

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7.8 Treatment in the Context of Mine Closure and Post Closure

The approach to mine drainage treatment during and after closure of mining operations must be placed in context with respect to the following factors:

- Changes in mine drainage flow and quality
- Climate change over the long term
- Long-term operations and maintenance
- Capital replacement cost
- Non-mining water user requirements
- Involvement from non-mining stakeholders

Mine drainage volumes requiring treatment may increase or decrease after mine closure. The opportunities for consumptive on-mine water usage decrease after closure, potentially resulting in increased excess mine drainage volumes. On the other hand, completion of rehabilitation work after closure may decrease the ingress of water into old mining operations, resulting in decreased excess mine drainage.

Management and support for long-term post-closure operation and maintenance of mine drainage treatment facilities may be limited. Passive treatment technologies are therefore considered more beneficial in the post-closure situation than active treatment technologies, where applicable.

Mine planners should consider post-closure water treatment system land requirements in the design of tailings storage facilities and mine waste dumps so that space is available, when needed, and post-closure water treatment does not become a major design constraint that forces the implementation of active treatment technologies. For example, a waste rock dump might be configured in a way that leaves adequate room at the toe for collection and passive treatment of residual seepage. A similar design protocol should be followed for tailings dams and other long-term mine waste facilities that may generate drainage in some cases in perpetuity.

The design life of post-closure treatment facilities should be based on geochemical model predictions of the long-term mine drainage flow and quality.

Replacement of capital infrastructure and equipment items must be taken into account for continued post-closure treatment. Mine drainage flows and associated pollutant loads are typically projected to continue for a considerable period after mine closure. In some cases, this long-term projection for continued treatment may even require a re-evaluation of the appropriate treatment approach and technology as research and technology development take place.

Communities and other non-mining economic activities may rely on the long-term availability of mine drainage. Such reliance is not necessarily negative because the transfer of mine drainage treatment facilities to a third party may assist in the sustainability of a post-mining situation. For instance, the Emalahleni Local Municipality in South Africa receives a substantial part of their drinking water supply from a mine water reclamation plant (Gunther et al., 2008).

The early involvement of non-mining stakeholders to identify and implement post-closure beneficial and economic use of mine drainage will assist in developing appropriate treatment infrastructure.

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7.9 Evaluation and Selection of Drainage Treatment Technologies

The evaluation of alternative drainage treatment technologies and the selection of an appropriate technology for a specific application require consideration of many of the following factors:

- Technical factors:
 - Scale of project
 - Location and accessibility of project
 - Location within the overall mine water cycle and circuits
 - Raw water composition and flow rate
 - Fit into the life cycle of the mine
 - Proven technology
 - Treated water quality requirements

- Reliable performance
- Risks related to implementation
- Operational factors:
 - Operations manpower and labour requirements
 - Process control and automation
 - Utility requirements (e.g., electrical power and water)
 - Chemicals and reagents requirements
 - Maintenance
 - Logistics and communications
- Environmental factors:
 - Residual impacts of treated water discharge
 - Climatic conditions
 - Waste disposal
 - Land use impacts
 - Regulatory approvals
- Financial factors:
 - Capital investment
 - Capital replacement costs
 - Operations and maintenance (O&M) costs
- Management factors:
 - Negotiating with regulators and other stakeholders
 - Defining decision process
 - Funding for all phases of mining
 - Negotiating for unexpected resources requirements
 - Maintaining companies' credibility and good standing
- Social Factors:
 - Community acceptance and involvement

A life cycle financial model approach is typically applied to evaluate the treatment project financial implications, including the following:

- Production and management of wastes and emissions
- Potential for by-product recovery
- Sustainability during active mining and post-closure phases

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7.10 Case Studies

The following case studies are provided to demonstrate some of the technologies highlighted in this chapter.

1. The Argo Tunnel - Pulsed Limestone Bed Treatment
2. Bisbee No. 7 stockpile – BioSulphide process
3. Equity Silver – High Density Sludge Treatment Plant
4. Keystone Mine – Constructed Wetlands

The Interstate Technology and Regulatory Council (ITRC) in the USA also has compiled several very useful case studies on their website:

http://www.itrcweb.org/miningwaste-guidance/case_studies.htm

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Reverse Osmosis Technology Helps Optimize Phosphate Mine Performance

Reverse osmosis technology helps optimize phosphate mine performance

by Chris Howell and Paul Hoeflerlin



PotashCorp – Aurora in North Carolina is the largest vertically integrated phosphate mining and chemical plant in the world. The facility produces sulfuric acid on site, which is reacted with the phosphate rock to produce phosphoric acid for use as feedstock for phosphate products.

PotashCorp's Aurora phosphate mine, in Beaufort County, NC, operates four low pressure boilers and four high pressure boilers to meet its process steam requirements. Maintaining these boilers is critical to meeting production demands.

Processing phosphate ore into phosphoric acid requires large amounts of sulfuric acid and steam. Ore is mined from the phosphate deposit with large bucket excavators after 9 to 30 m (30 to 100 ft) of overburden is removed. The phosphate ore is mixed with water to make a slurry that is pumped to the mill. Elemental sulfur is burned in the presence of air to make sulfuric acid. This process is highly exothermic and boilers produce steam while cooling the process. The phosphate ore is reacted with sulfuric acid and the resulting products are refined into different grades of phosphoric acid for fertilizers, feed stock and food additives.

Plant operators at the mine were challenged with boiler feed water that was high in silica and other contaminants. As a result, boiler operations were experiencing difficult, labor-intensive operations and equipment deterioration. This resulted in higher operating costs and diminished processing performance. The facility was operating an aged boiler feedwater pretreatment system that included

warm lime softening followed by pressure filtration and a cation/anion/mixed bed demineralizer system. The demineralizer required frequent regeneration and chemical usage was high and costs were difficult to predict.

Depressurized well water (DPW water), used for plant utility water and boiler makeup, is very high in silica, in the range of 60-70 ppm. Silica can form scale at pressures below 600 psig. Above 600 psig, silica starts to volatilize, carrying over with steam to potentially form deposits on downstream equipment and processes.

Mine officials decided to upgrade the boiler feedwater pretreatment system to a state-of-the-art reverse osmosis (RO) membrane system. This new system, which started up in March 2009, has improved operating performance and reduced water consumption and overall

operating expenditures.

World's largest

The Aurora phosphate operation is a subsidiary of the Potash Corporation of Saskatchewan (PotashCorp). The mine is the largest vertically integrated phosphate mining and chemical plant in the world. The facility produces sulfuric acid on site, which is reacted with the phosphate rock to produce phosphoric acid for use as feedstock for phosphate products. The operation has an annual capacity of 6 Mt/a (6.6 stpy) of phosphate rock, 1.2 Mt/a (1.3 million stpy) of phosphoric acid and 181 kt/a (200,000 stpy) of feed phosphate. Steam is generated as part of the sulfuric acid production process. It is also used in the generation of electrical power for both the plant and resale.

To ensure the new pretreatment system would meet boiler feed water requirements, mine officials teamed with Crown Solutions, a business unit of Veolia Water Solutions & Technologies, to conduct testing using a pilot trailer custom built for this specific application. The 5.7-L/min (1.5-gpm) pilot reverse osmosis system was scaled to the project, to precisely match the hydraulics of the proposed full-scale system. Operating at only 5.7 L/min (1.5 gpm), the pilot system was designed with the same hydraulic flow rates across the multimedia filters and the softeners as the proposed full-scale system.

During initial testing, the pilot unit's silt density

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Table 1

Specific water contaminants that are present at the beginning of each process, and what remains after processing.

index (SDI) test apparatus lost flow within five minutes of operating due to particulate fouling, and membrane performance showed poor results. It was subsequently determined that the raw process water contained very fine particulates (98 percent less than 2-micron) that flowed through the multimedia filters and softeners, fouling the RO membranes.

Once a specialized blend of coagulants was developed and used, the pilot system significantly lowered SDIs to the 0.5 range. Reverse osmosis with an SDI below 3 is considered acceptable, so an SDI of 0.5 was a significant improvement.

The final results of the eight-week pilot test showed good filtration results, good operation of the water softener and good operation of the membrane system. The RO system produced high quality water, with the operation showing no signs of fouling or scaling. Based on the pilot results, the mine gave the green light for the installation of the full-scale RO system.

System overview

The new boiler feed water pretreatment system includes multimedia filtration (MMF), ion exchange softening and reverse osmosis. DPW water for plant utility water and boiler makeup is pumped out of the mine from deep wells at roughly 145 L/sec (2,300 gpm).

Upstream of the multimedia filtration, coagulant and sodium hypochlorite are fed into the raw water at two surge tanks. These are converted, old warm lime softening vessels with about 76 L/sec (1,200-gpm) nominal capacity each. Sodium hypochlorite is used as a biocide to destroy any biological contaminants in the feed water and minimize the ability for microorganisms to grow within and contaminate the water treatment system.

Water from the surge tanks is then pumped to five 2.4-m (96-in.) multimedia filters with a flow capacity of 29 L/sec (460 gpm) per filter. The down-flow multimedia filtration system is designed to remove total suspended solids (TSS) from the DWP makeup water. All filters run simultaneously in automatic mode. When this system goes into a backwash sequence, RO reject water from a backwash water storage tank is used for backwashing. The use of RO reject for multimedia filter backwashing optimizes water usage and minimizes the virtual water footprint.

The filtered water flows to the softener system, which includes seven softeners, each with a flow rate of 21 L/sec (328 gpm). The system is designed to automatically manage the number of softening units online, based on the number of reverse osmosis units online in standard automatic operation. During regeneration, brine

Multi-media filters		
	Influent	Effluent
Turbidity (NTU)	8	<1
SDI ₁₅	>6	<1
Conductivity (uS/cm)	650	650
Hardness (ppm, as CaCO ₃)	450	450
1) Primarily for TSS removal 2) Addition of coagulant to aid in particulate removal 3) Addition of sodium hypochlorite to kill biological		
Ion exchange softeners		
	Influent	Effluent
Turbidity (NTU)	<1	<1
SDI ₁₅	<1	<1
Conductivity (uS/cm)	650	670
Hardness (ppm, as CaCO ₃)	450	<1
1) Units regenerated with a 12% brine solution 2) Serves as a form of ion exchange; NOT removal		
First pass RO machine		
	Influent	Effluent
Turbidity (NTU)	<1	<1
SDI ₁₅	<1	<1
Conductivity (uS/cm)	650	10
Hardness (ppm, as CaCO ₃)	<1	0
1) Antiscalant is injected upstream of this system 2) Bisulfite is injected as a means of biological control and to bind any remaining free chlorine		
Second pass RO machine		
	Influent	Effluent
Turbidity (NTU)	<1	<1
SDI ₁₅	<1	<1
Conductivity (uS/cm)	10	<1*
Hardness (ppm, as CaCO ₃)	0	0
* Heavily dependent on amount of CO ₂ in feedwater		

is injected into the system to reverse the ion exchange process. The system includes the ability to use brine reclaim where a significant portion of the brine waste is recaptured and sent to a brine reclaim tank to be used in future regenerations.

Reverse Osmosis



The softener system includes seven softeners, each with a flow rate of 21 L/sec (328 gpm). The system is designed to automatically manage the number of softening units online. During regeneration, brine is injected into the system to reverse the ion exchange process.

This lessens the amount of virgin brine required for the process and further reduce the water needed to operate the process.

The softening resin is also a specialized resin — a shallow shell technology (SST) resin — by Purolite resin company. The polystyrene resin keeps ion exchange sites to the outer 60 percent of the resin beads, which allows for more efficient regenerations and lower salt usage compared with other more conventional resin types.

The RO system is designed to operate as a single and double pass system. All four RO units have a 16:8 array and 144 membranes. Units A, B and C are single pass units. They are always fed with soft water from the upstream softeners. Unit D can operate in both first pass mode (fed water from the softeners) or second pass mode. If Unit D acts as a second pass unit, permeate off the first pass feeds the second pass and produces high quality water.

The concentrate/reject of the units operating in the first pass mode, which is highly concentrated in TDS, is sent to waste. This reject is also captured in the backwash tank for the MMF/softener backwash. The concentrate of Unit D, if operating in second pass mode, is recovered and sent to the first pass RO feed.

RO units A, B and C, acting as first pass, have a designed recovery rate of 70 percent. RO Unit D, when acting in second pass mode, has a designed recovery rate of 90 percent. A major portion of the first pass water is process water used in Potash Corp-Aurora's low-pressure boilers. Second pass RO water is used for high purity processes and in the plant's high-pressure boiler. In addition to filter and softener backwash, RO reject water can be used for softener regeneration and cooling tower makeup.

In effect, Potash Corp-Aurora operates a two-stage RO in which two RO systems are running in series with the permeate of the first acting as the

feed to the second. Staged or series RO operation is typically done when a single-stage RO system does not produce the required quality of product water. For Potash Corp-Aurora, two-stage RO is justified because the additional expenses of operating the second RO system is lower than alternative forms of polishing the first-stage RO permeate to reach a higher quality of final product water.

The first pass RO machine produces a permeate stream and a concentrate stream. The magnitude of these flows depends on the recovery of the RO system. Running at 70 percent recovery, the first pass RO reject stream is 3.33 times concentrated versus the feed water. Since the silica in the feed water is 60 to 70 ppm, this means that the silica in the first pass RO reject stream is 200 to 234 ppm. Based on the solubility of silica, this could cause a scaling issue in the concentrate stream of the RO machine.

A key issue regarding the success of the project was based on developing an anti-scalant blend that could handle the high silica levels. A silica anti-scalant program using this blend was implemented to keep the membranes at optimal performance. Based on the chemistry of the water, it was expected that the membranes would require cleaning every three months. In the initial operation of the system, four months passed before the first cleaning was required. The initial cleaning of the RO membranes restored full flow to the system, thereby indicating the anti-scalant program has been successfully implemented and is working well.

Improved performance

The new boiler water pretreatment system provides the PotashCorp-Aurora's facility several advantages over its previous system. Although the initial investment in the new technology was significant, there are substantial savings in maintenance, labor costs and chemicals. There are also the additional benefits of increased reliability and availability of the water plant, as well as the savings due to their extended project life.

The new RO system at the Aurora mining complex has consistently provided high quality feed water, ensuring greater reliability of its systems. It has also proven its sustainability and worth in several other ways versus the performance of the old system.

The RO technology now offers Aurora's water treatment operators an easier system to operate. It has significantly reduced maintenance demands at the plant and lowered the overall cost of boiler feed water operations. High quality boiler feed water is helping to ensure the maximum life of the boilers as well as optimal thermal performance to help the plant operate more efficiently. ■

PolyMet Reports Successful Water Treatment Pilot Plant



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NEWS RELEASE

2012-14

POLYMET REPORTS SUCCESSFUL WATER TREATMENT PILOT PLANT

St. Paul, Minnesota, October 10, 2012 - PolyMet Mining Corp. (TSX: POM; NYSE MKT: PLM) ("PolyMet" or the "Company") is pleased to announce that it has successfully treated over one million gallons of water through its Reverse Osmosis (RO) pilot water treatment plant. The pilot plant has been operational since May 2012 and is designed to treat water containing elevated levels of sulfate. The purpose of the pilot plant is to demonstrate the ability to comply with Minnesota's strict water quality standards using a modular system that can be expanded to a full-scale treatment plant.

PolyMet owns 100% of the NorthMet Project, which comprises the development-stage NorthMet copper-nickel-precious metals ore-body and the nearby Erie Plant, located near Hoyt Lakes in the established mining district of the Mesabi Iron Range in northeastern Minnesota.

The NorthMet Project's potential impact on water quality is a key issue in the environmental review and permitting process. The successful pilot plant test is a significant accomplishment, demonstrating that water with elevated sulfate levels can be treated to meet Minnesota's 10 parts-per-million sulfate standard for waters used for the production of wild rice.

PolyMet partnered with GE Water & Process Technologies (GE) and Barr Engineering to design and operate the pilot plant using Reverse Osmosis (RO) membrane technology developed by GE. The test work demonstrates the technical and regulatory viability of Reverse Osmosis as a water treatment method that will enable PolyMet to successfully develop the NorthMet copper-nickel deposit and meet state and federal water quality standards.

"Our ability to successfully demonstrate the effectiveness of our planned water treatment systems prior to permitting and construction is extremely important," said Jon Cherry, President and CEO of PolyMet. "We want both the community and agencies to understand our commitment to constructing and operating NorthMet Project in an environmentally responsible manner that meets all applicable standards."

Yuvbir Singh, General Manager, Engineered Systems, GE Water and Process Technologies stated, "GE is pleased to be working with PolyMet and Barr to develop this site specific water treatment system. The combination of PolyMet and Barr's design and operating knowledge with GE's worldwide expertise in membrane treatment technology shows how application of the right technology can achieve compliance with very strict water quality standards."

PolyMet continues to operate the pilot treatment plant to refine operating parameters and increase efficiencies.

* * * * *

About PolyMet

PolyMet Mining Corp. (www.polymetmining.com) is a publicly-traded mine development company that owns 100% of Poly Met Mining, Inc., a Minnesota corporation that controls 100% of the NorthMet copper-nickel-precious metals ore body through a long-term lease and owns 100% of the Erie Plant, a large processing facility located approximately six miles from the ore body in the established mining district of the Mesabi Range in northeastern Minnesota. Poly Met Mining, Inc. has completed its Definitive Feasibility Study and is seeking environmental and operating permits to enable it to commence production. The NorthMet project is expected to require approximately two million hours of construction labor, creating approximately 360 long-term jobs, a level of activity that will have a significant multiplier effect in the local economy.

POLYMET MINING CORP.

Per: "Jon Cherry"

Jon Cherry, CEO

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This news release contains certain forward-looking statements concerning anticipated developments in PolyMet's operations in the future. Forward-looking statements are frequently, but not always, identified by words such as "expects," "anticipates," "believes," "intends," "estimates," "potential," "possible," "projects," "plans," and similar expressions, or statements that events, conditions or results "will," "may," "could," or "should" occur or be achieved or their negatives or other comparable words. These forward-looking statements may include statements regarding our beliefs related to exploration results and budgets, reserve estimates, mineral resource estimates, work programs, capital expenditures, actions by government authorities, including changes in government regulation, the market price of natural resources, costs, ability to receive environmental and operating permits, job creation, or other statements that are not a statement of fact. Forward-looking statements address future events and conditions and therefore involve inherent known and unknown risks and uncertainties. Actual results may differ materially from those in the forward-looking statements due to risks facing PolyMet or due to actual facts differing from the assumptions underlying its predictions.

PolyMet's forward-looking statements are based on the beliefs, expectations and opinions of management on the date the statements are made, and PolyMet does not assume any obligation to update forward-looking statements if circumstances or management's beliefs, expectations and opinions should change.

Specific reference is made to PolyMet's most recent Annual Report on Form 20-F for the fiscal year ended January 31, 2012 and in our other filings with Canadian securities authorities and the U.S. Securities and Exchange Commission, including our Report on Form 6-K providing information with respect to our operations for the three months ended July 31, 2012 for a discussion of some of the risk factors and other considerations underlying forward-looking statements.

The TSX has not reviewed and does not accept responsibility for the adequacy or accuracy of this release.

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Membrane Selection and Optimal Ammonium Nitrate Chemistry for Reverse Osmosis Treatment of Explosive Wastewater

By: Gary Davis, David Paulson, John Delebo

Jan. 1, 1990

☆☆☆☆☆ (0 votes)

A process for manufacturing explosives produces an ammonium nitrate-laden wastewater, which can be successfully treated with reverse osmosis (RO). Key to RO use is balancing the competing waste solution chemistry requirements of the two target solutes, to enhance the performance of the particular RO membrane chosen. While the EPA requires only the reduction of the NO₃ and NH₄⁺ discharge levels one manufacturer has chosen the approach of reusing both permeate and concentrate streams to 'close the loop.' A lab-scale test followed by field-site testing were performed to evaluate membrane choices and optimum solution chemistry.

The solution pH directly affects the rejection of both the ammonium and the nitrate ions. A solution pH above 7 has a very negative impact on ammonium ion rejection, since the equilibrium between the monovalent ammonium ion and the uncharged ammonia molecule shifts toward ammonia as the pH approaches the basic range. With little ionic charge to enhance rejection, the very small ammonia molecule readily permeates RO membranes. This effect was demonstrated during the pilot testing, where as the pH was raised from 5.0 to 8.5 the ammonia rejection of the RO system dropped from 87% to 62%.

Share

In contrast, nitrate ion rejection is improved by increasing the pH. During the lab test, adjustment of the solution pH from 2.0 to 4.1 improved nitrate rejection from 31% to 94%. The mechanism for this improvement is not as obvious, but probably also involves the effect on the apparent charge of the ion. Decreasing the pH shifts the equilibrium away from the monovalent nitrate ion toward an uncharged nitric acid molecule (HNO₃). Again, such a small uncharged molecule more readily permeates the RO membrane.

Previous experience has demonstrated that, while the NH₄⁺ rejection level is similar for the membranes of both an aromatic polyamide (PA) and a cellulose acetate (CA) blend (di- and triacetates), the PA membrane exhibits substantially better rejection of the NO₃⁻ ion. To simplify, this can be attributed to the inherent difference in surface chemistry characteristics between the membrane polymers, which require smaller pores in the PA membrane - relative to CA - to achieve similar NaCl rejection levels (NaCl is the most common marker solute to commercially rate RO membranes). This pore size difference is demonstrated by the superior small organic molecule rejection capability of PA membranes relative to CA.

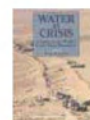
In this case, the 'fluffy' charge density of the NO₃⁻ ion (due to resonance) reduces the effect of the dielectric repulsion mechanism (as defined by Sourirajan) and rejection becomes more dependent on pore size. Hence, lower rejection values are obtained for the CA membrane with its larger pores. In contrast, the NH₄⁺ ion, with a point charge density behavior more like typical salt ions, displays similar rejection levels for both membrane polymers.

Initial lab testing included a membrane scan and waste stream chemistry manipulation. An aromatic polyamide RO membrane was selected for field-site testing on the condensate from a nitric acid neutralization process. Varying the pH from 2.9 to 8.2 identified a technically acceptable range of 3.0 to 7.0. Economic considerations narrowed the optimum processing

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