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REFERENCE GUIDE
to Treatment Technologies for Mining-Influenced Water

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Acronyms and Abbreviations

ALD	Anoxic Limestone Drain
ARD	Acid Rock Drainage
BCWTP	Bingham Canyon Water Treatment Plant
BCR	Biochemical Reactor
C	Celsius
CPFM	Colloid Polishing Filter Method
ED	Electrodialysis
EDR	Electrodialysis Reversal
FBR	Fluidized Bed Reactor
FF	Filter Flow
FRTR	Federal Remediation Technologies Roundtable
GARD	Global Acid Rock Drainage
GPD	Gallons per Day
GPM	Gallons per Minute
ITRC	Interstate Technology and Regulatory Council
L/min	Liters per Minute
µg/L	Micrograms per Liter
mg/L	Milligrams per Liter
MGD	Million Gallons per Day
MIW	Mining-Influenced Water
ND	Not Detected
OLC	Open Limestone Channels
OU	Operable Unit
O&M	Operation and Maintenance
PRB	Permeable Reactive Barrier
RCTS	Rotating Cylinder Treatment System
SAPS	Successive Alkalinity Producing Systems
SME	Silica Micro Encapsulation

TCLP	Toxicity Characteristic Leaching Procedure
TDS	Total Dissolved Solids
USD	United States Dollars
EPA	United States Environmental Protection Agency
ZVI	Zero Valent Iron

Notice and Disclaimer

This report compiles information on selected technologies for treatment of mining-influenced water (MIW). The report includes those technologies that are used to treat MIW for which information was readily available. It is not a comprehensive review of all current technologies. The report contains information for interested stakeholders, including governments, the public and the regulated community. The report does not provide guidance regarding the selection of a specific technology, and the use of a specific technology may or may not result in compliance with applicable federal, state or tribal environmental requirements, even if water quality improves as a result of using a given treatment technology.

The mention of trade names, specific vendors or products does not represent an actual or presumed endorsement, preference or acceptance by the United States Environmental Protection Agency (EPA) or the federal government. Stated results, conclusions, usages or practices do not necessarily represent the views or policies of EPA.

This report has undergone EPA and external review by experts in the field. However, information in this report comes from many sources, some of which have not been peer reviewed. EPA recommends that users refer to applicable regulations, policies and guidance documents regarding selection of cleanup remedies and implementation of mine water treatment and mine cleanup actions; selected references and additional resources are provided herein. The Agency notes that this report may be revised periodically without public notice. EPA welcomes public comments on this report at any time and will consider those comments in any future revisions of the document.

Representatives from various EPA offices provided assistance with development of this report and internal peer review. They include Bill Adams, Robin Anderson, Diana Bless, Barbara Butler, Bette Conway, Anne Dailey, Elisabeth Freed, Gregory Gervais, Mike Gill, Doug Grosse, Jim Hanley, Ed Hathaway, Gary Hudiburgh, Stephen Hoffman, Joy Jenkins, Tom Kady, Michelle Kerr, Kira Lynch, Shahid Mahmud, Krista McKim, Carlos Pachon, Mark Purcell, Elaine Suriano, Dave Tomten, Clifton Townsend, Stuart Walker and Andy Zownir. Additional external reviews were provided by: Amanda Aspatore, National Mining Association; Tim Buxton, USDA Forest Service; Mark Fitch, Missouri University of Science and Technology; Gary Hickman and Jim Stefanoff, CH2M Hill; Ann Maest; David Reisman, CDM Smith; Brett Waterman, Freeport McMoRan Copper & Gold; and Paul Ziemkiewicz, West Virginia University.

EPA's Office of Superfund Remediation and Technology Innovation prepared this report. For further information about this report, please contact Michele Mahoney at EPA's Office of Superfund Remediation and Technology Innovation, at (703) 603-9057, or by email at mahoney.michele@epa.gov.

Introduction

This report provides an overview of select mining-influenced water (MIW) treatment technologies used or piloted as part of remediation efforts at mine sites. The report is intended to provide information on treatment technologies for MIW to federal, state and local regulators, site owners and operators, consultants, and other stakeholders. The technologies described in this report are applicable to treatment of water from both coal and hard-rock mine operations. The report provides short descriptions of treatment technologies and presents information on the contaminants treated, pre-treatment requirements, long-term maintenance needs, performance and costs. Sample sites illustrate considerations associated with selecting a technology. Website links and sources for more information on each topic are also included.

MIW is defined as any water whose chemical composition has been affected by mining or mineral processing and includes acid rock drainage (ARD), neutral and alkaline waters, mineral processing waters and residual waters. MIW can contain metals, metalloids and other constituents in concentrations above regulatory standards. MIW affects over 10,000 miles of receiving waters in the United States, primarily by acidic drainage.[1]

EPA is evaluating more cost-effective and lower-maintenance treatment systems to decrease the costs and improve the efficiency of mine site cleanups. Hence, this report focuses on passive treatment methods, but also includes recently developed or not widely utilized active treatment systems and passive-active hybrid systems. The report does not include all traditional active technologies, such as lime precipitation or high-density sludge systems. Resources that summarize these and other technologies include the Interstate Technology and Regulatory Council (ITRC) Mining Waste Treatment Selection technology overview webpage (http://www.itrcweb.org/miningwaste-guidance/technology_overviews.htm) and the Network for Acid Prevention's Global Acid Rock Drainage (GARD) Guide (http://www.gardguide.com/index.php/Main_Page).

EPA's policy for reducing the environmental footprint of activities used to clean up contaminated sites is based on core element principles of reducing energy usage, air pollution, and impacts on water resources, improving waste management, and protecting ecosystem services. There are significant opportunities to reduce the environmental footprint associated with characterizing MIW and using passive treatment systems. EPA's green remediation best management practices for treating MIW can be found at http://cluin.org/greenremediation/docs/GR_factsheet_miningsites.pdf.

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[1] ITRC, 2008.

In recent years, development and implementation of passive systems has increased. However, additional pilot studies and case studies are needed to assess their effectiveness. With time, EPA expects that the pool of technology options will expand and shift away from high-energy-use, high-maintenance systems to low-energy-use, low-maintenance systems.

Active or passive methods can remove or reduce the concentration of contaminants in MIW. The International Network for Acid Prevention's GARD Guide considers active treatment as technologies that require ongoing human operation, maintenance and monitoring, and have or use external sources of energy, infrastructure and engineered systems. Passive treatment refers to processes that do not require frequent human intervention, operation or maintenance, and that typically employ natural construction materials (e.g., soils, clays, broken rock), natural treatment media (e.g., plant residues such as straw, wood chips, manure, compost), and promote growth of natural vegetation.[2] Passive treatment systems use gravity flow for water movement, and passive energy sources such as solar or wind power. In some arid climates, they might also include the use of evaporation or infiltration. Both active and passive treatment methods potentially combine physical, biological and chemical approaches to treat MIW. The main purpose of both classes of technologies is to raise pH, lower dissolved metal concentrations, and lower sulfate. Active or passive treatment of MIW generally requires long-term maintenance and funding.

Key factors when considering MIW treatment technologies include the amount of available land surface and its topography; system longevity and maintenance needs; flow rate and strength; site accessibility and remoteness; availability of utilities (especially power sources); performance criteria; design, capital, and operation and maintenance costs; and climate impacts on system effectiveness. Treatment systems typically include multiple steps of treatment with more than one technology. Therefore, the associated costs of constructed systems are higher than the costs included in this report for individual technologies. Although cost information is not available for all technologies, the U.S. Office of Surface Mining has developed an online program for evaluating cost of treatment methods, called AMDTreat. The program is available at http://amd.osmre.gov/ and is designed to predict approximate costs for various sets of treatment steps.

Methodology

EPA identified the MIW treatment technologies in this report by reviewing technical literature, including EPA reports and EPA databases such as the Federal Remediation Technologies Roundtable (FRTR: www.frtr.gov) and EPA's Clean-Up Information (CLU-IN) website (www.clu-in.org), and by contacting subject matter experts. Research included results of

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[2] GARD, 2009.

demonstration projects conducted by EPA’s Superfund Innovative Technology Program and the joint EPA and Department of Energy Mine Waste Technology Program. MIW treatment experts reviewed this report. However, due to limited availability of relevant peer-reviewed publications, some of the references cited may not have been peer reviewed. Examples of such sources include site-specific reports and industry publications. In addition to the technologies discussed in this report, engineering controls and source controls are important parts of managing MIW. Source control strategies are outside the scope of this report.

MIW treatment technologies were identified and divided into active and passive approaches. For each technology, the report summarizes the following information, where available: technology description; technology operations; constituents treated; sites where the technology was applied at full, pilot or bench scale; long-term maintenance requirements; system limitations; system costs; and treatment effectiveness. The technologies included in the report are not a comprehensive list of all treatment options. Based on the available information, EPA reviewed the types of waste and contaminants treated and summarized the results from use of each technology. Performance data are reported based on influent and effluent concentrations. For many of the technologies described in this report, there were gaps in the available information. EPA did not perform independent evaluations of technology performance in developing this report.

The report provides detailed information on 16 technologies. These and additional technologies are compiled in the Appendix A table. For each technology, the table provides information on treated constituents, technology scale, example sites, operations, long-term maintenance requirements, system limitations, costs, and effectiveness.

EPA will continue to evaluate MIW treatment technologies and solicit input on new developments. EPA intends to build on this report by developing strategies for implementing and expanding the use of passive treatment technologies.

Technology: Anoxic Limestone Drains

Technology Description

Anoxic limestone drains (ALDs) involve the burial of limestone in oxygen-depleted trenches. MIW is conveyed into these trenches. ALDs generate alkalinity and must be followed by a unit such as an aeration cascade, pond or aerobic wetland that oxidizes and removes the precipitated metals. Limestone is a low-cost and effective way to generate alkalinity. However, it must be used in appropriate conditions to ensure its effectiveness.

Constituents Treated

ALDs treat acidity.

Operations

An ALD consists of a trench containing limestone encapsulated in a plastic liner that is covered with clay or compacted soil (Figure 1). Surrounding the limestone with an impervious plastic liner also helps maintain anoxic conditions in the drain. The soil used to cover the plastic must consist of finely graded materials and be compacted to limit oxygen diffusion to the drain. The anoxic conditions will prevent the oxidation of ferrous iron to its ferric state, which oxidizes readily to ferric hydroxide in the presence of oxygen at pH values greater than 3.5. The cap also prevents water infiltration and helps prevent carbon dioxide from escaping.

Figure 1: ALD Cross-Section

Prior to development and installation of an ALD, influent water must be characterized to ensure effective system design. This includes looking at seasonal variations. In addition to flow rate, important influent characteristics include dissolved oxygen content, acidity and alkalinity, ferric and ferrous iron concentrations, and aluminum concentrations. If the pH is less than 5, iron concentrations should be speciated into ferric and ferrous. If ferric iron is present, the water may need to be pretreated to reduce the ferric to ferrous ion. This reduction step is necessary because ferric iron will precipitate and armor the limestone within the ALD, resulting in reduced efficiency. At pH levels above 5, generally dissolved iron is in the ferrous form. Water quality data will determine the applicability of an ALD, and flow data will provide sizing criteria for the ALD.

Skousen (1991) found that high-grade limestone with $CaCO_3$ content of greater than 90 percent should be used.[3] At this percentage, the limestone dissolves quickly, can obtain high alkalinity levels in the water (near 300 milligrams per liter (mg/L) as $CaCO_3$), and has fewer impurities that could clog drains. Limestone used in ALDs is usually in the form of pebbles or rocks, with a particle spectrum ranging from 1.5 to 4 inches. Small-size particles provide more surface area for more rapid dissolution and alkalinity generation, while the larger-size particles will dissolve more slowly and provide system longevity, and maintain distributed water movement through the drain to limit short circuiting. A mixture of sizes will help ensure continuous flow through the drain.

ALDs can be installed in remote areas due to their passive nature and the fact that utilities are not required for implementation. ALDs may also be used to treat a wide range of MIW flow rates. About 15 hours of contact time is necessary to achieve a maximum concentration of alkalinity.[4] To achieve 15 hours of contact time within an ALD, 2,800 kilograms of limestone are required for each liter per minute (L/min) of peak MIW flow.[5] Therefore, ALD design typically involves calculating the size and mass needed to create an effective system based on the flow rate.[6]

Figure 2: Anoxic Limestone Drain Outflow at the Midwestern Reclamation Site, Pike County, Indiana

Image Source: http://igs.indiana.edu/Reclamation/Reclamation.cfm

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[3] Skousen, 1991.
[4] Watzlaf, 2004.
[5] Watzlaf, 2004.
[6] ITRC, 2010.

Once the water exits the drain, sufficient area must be provided for metal oxidation, hydrolysis and precipitation to occur. Settling basins or ponds can be used for this purpose. State water regulations governing alkalinity, pH, dissolved oxygen and hardness discharge limits should be taken into account during ALD system design. Figure 2 above shows the vegetated cap of an ALD in the background (see Figure 1 for an ALD cross-section diagram) and the outflow of treated water.

Table 1: Examples of ALD Implementation

Valzinco Mine, Virginia Mine Type: Underground lead/zinc/copper mine http://www.itrcweb.org/miningwaste-guidance/cs18_valizinco_mine.htm.	**Copper Basin Mining, Tennessee** Mine Type: Underground and surface copper mines http://www.itrcweb.org/miningwaste-guidance/cs2_copper_basin.htm.
Hartshorne/Whitlock-Jones, Hartshorne, Oklahoma Mine Type: Underground coal mine http://www.itrcweb.org/miningwaste-guidance/cs13_hartshorne.htm.	**Ohio Abandoned Bituminous Coal, Ohio** Mine Type: Underground and surface coal mines http://www.itrcweb.org/miningwaste-guidance/cs1_se_ohio.htm.
Tecumseh - AML Site 262, Indiana Mine Type: Surface coal mine http://www.itrcweb.org/miningwaste-guidance/cs37_tecumseh.htm.	**Tennessee Valley Authority, Alabama** Mine Type: Underground coal mine http://www.itrcweb.org/miningwaste-guidance/cs33b_tva_alabama.htm.

Long-Term Maintenance

The limestone will need replenishment once depleted.[7] Factors considered in the sizing of an ALD include the levels of dissolved metals present in the mine drainage and the retention time needed to raise the pH, as well as the amount of area available for construction.[8] Dimensions are typically about 1 meter deep, 1 to 7 meters wide, and 25 to 100 meters long, with the exact dimensions of a system depending on the retention time desired, the influent flow rate and metals concentrations, the purity of the limestone, discharge limits, and the length of time desired for system longevity.[9] Capping a system with topsoil and vegetation provides protection against erosion.

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[7] ITRC, 2010.

[8] ITRC, 2010.

[9] Watzlaf, 2000a.

System Limitations

Excessive iron and aluminum precipitation in an ALD will reduce the effective lifetime of the bed by armoring the stone, resulting in reduced permeability and calcite dissolution rates. Metal removal must occur elsewhere in the treatment process to prevent clogging of the limestone bed and premature failure. The buildup eventually clogs the open pore spaces, resulting in abnormal flow paths that can reduce both the retention time of MIW in the ALD and the reactive surface area of the limestone.

Longevity of treatment is also a concern for ALDs in terms of water flow through the limestone. Faulkner and Skousen (1994) and Watzlaf et al. (1994) have observed clogging of pores with precipitated iron and aluminum hydroxides. The addition of calcium from limestone dissolution can cause the precipitation of gypsum in MIWs with sulfate concentrations greater than 1,500 mg/L.[10] Most ALD systems exhibit reduced effectiveness over time and eventually require maintenance or replacement.

Although ALDs are documented to successfully raise the pH of MIWs, the chemical characteristics of the influent mine water can cause variations in alkalinity generation and metal removal.

ALDs are suitable for treatment of MIW with low concentrations of ferric iron, dissolved oxygen and aluminum. However, when any of these three parameters are elevated, there can be armoring of limestone, which will slow its dissolution rate because the surface becomes coated. The buildup of iron and aluminum hydroxides can clog pore spaces, decrease retention times, and reduce the reactive surface area of the limestone.

Costs

Constructing and operating ALDs to treat MIW can be cost effective. Specific constituents in the water determine ALD installation costs. A typical ALD built at most locations in Canada costs an estimated $6,000 to $37,000 (2013 USD), depending on chosen dimensions and design flow.[11] The ITRC reported capital costs at an abandoned mine in Alabama of about $0.27 per 1,000 gallons of water. Operation and maintenance costs were about $0.11 per 1,000 gallons of treated water.[12]

Effectiveness

The success of an ALD depends on site-specific conditions, primarily on low dissolved oxygen, and minimal ferric iron and aluminum concentrations in the drainage. Where these conditions exist and the drains are properly constructed and maintained, a service life of 25 to 30 years is anticipated. Watzlaf et al (2000b) assessed the long-term performance of 10 ALDs used for

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[10] Nairn, 1991.

[11] MEND, 1996.

[12] ITRC, 2010.

treating coal mine drainage by analyzing a decade of influent and effluent water data. The authors found that alkalinity in the effluent ranged from 80 to 320 mg/L as $CaCO_3$, with levels near maximum reached after about a 15-hour detention time in the ALD. Consistent alkalinity values were achieved for over 10 years in ALDs where influent contained less than 1 mg/L of ferric iron and aluminum.

Factors affecting the level of alkalinity include contact time, the influent partial pressure of carbon dioxide, influent pH, the particle size of the crushed limestone, the $CaCO_3$ content of the limestone and the initial calcium concentration in the drainage.[13] Contact time and the partial pressure of carbon dioxide significantly affect the final concentration of alkalinity.[14]

Under the right circumstances and if constructed properly, limestone drains can add up to 300 mg/L alkalinity as $CaCO_3$ to water.[15] The amount of generated alkalinity is limited by the dissolution rate of limestone in water. As pH decreases, limestone is more soluble and higher amounts of alkalinity can be generated. In the absence of dissolved ferric iron and aluminum, ALDs have continued to perform well with no significant seasonal variations or long-term reduction in effectiveness.

Table 2 displays the treatment efficiencies observed at 10 ALD sites. All of the sites are located in Pennsylvania, except the Elklick site, which is located in Maryland.

Table 2. Examples of ALD Efficiencies

ALD	Acidity[1], mg/L as $CaCO_3$		Alkalinity, mg/L as $CaCO_3$		Calcium (mg/L)		pH		Sulfate (mg/L)	
	In	Out	In	Out	In	Out	In	Out	In	Out
Howe Bridge 1	472	352	32.6	155	115	223	5.74	6.30	1,319	1,314
Howe Bridge 2	411	274	35.3	163	157	209	5.40	6.48	1,210	1,211
Elklick	52	-63	33.8	159	258	232	6.06	6.73	334	327
Jennings	280	-33.5	0	139	ND	201	3.23	6.16	633	620
Morrison[2]	387	51.4	28.7	278	82.9	208	5.19	6.35	1,256	1,016
Filson – R[2]	100	-139	47.9	299	69.2	180	5.73	6.49	408	438
Filson – L[2]	104	-175	47.9	317	77.1	129	5.73	6.60	408	395
Schnepp[3]	307	-42.5	0	168	69.2	189	3.28	6.17	980	745
REM-R[3]	1,148	835	0	54	258	206	4.28	5.45	2,825	2,394
REM-L[3]	ND	259	ND	113	ND	198	ND	6.00	ND	1,256

Source: http://www.imwa.info/bibliographie/19_2_098-110.pdf.
Negative acidity values indicate that there is measurable alkalinity.
"In" concentrations based on water quality of a nearby seep.
"In" concentrations based on historical water quality data from untreated mine drainage prior to ALD construction. Influent values are not available for REM-L.
ND = Not Determined

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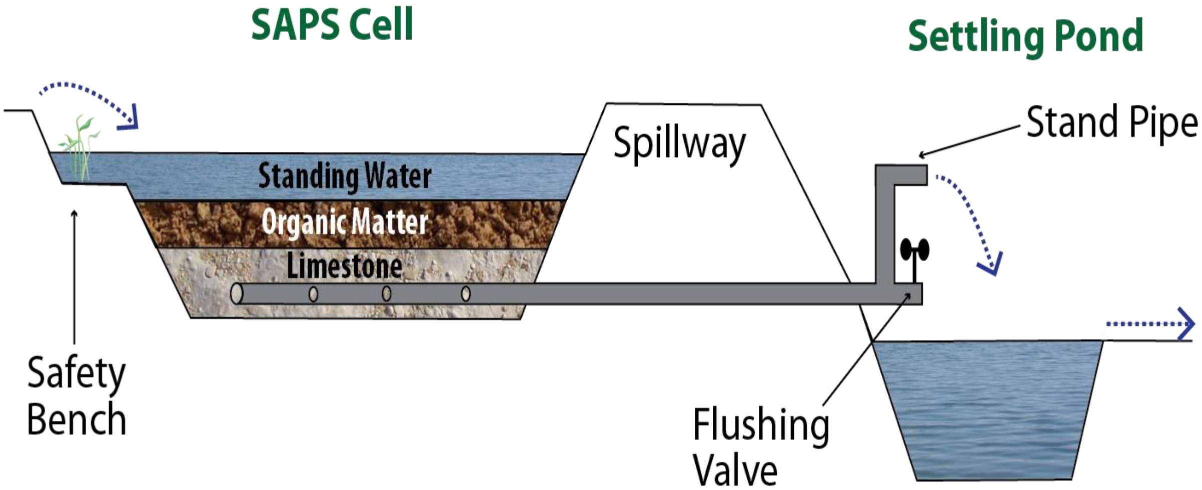
[13] Watzlaf, 2000b.
[14] Hedin, 1994.
[15] Watzlaf, 2004.

Technology: Successive Alkalinity Producing Systems (SAPS)

Technology Description

Successive alkalinity producing systems (SAPS) combine an ALD and a permeable organic substrate into one system that creates anaerobic conditions prior to water contacting the limestone.[16] A SAPS contains a combination of limestone and compost overlain by several feet of water (Figure 3). Mine drainage enters at the top of the pond, flows down through the compost where the drainage gains dissolved organic matter and becomes more reducing, and then flows into the limestone below, where it gains alkalinity.[17] Dissolution of the limestone raises the pH of the water, resulting in the precipitation of aluminum, copper and iron. The precipitated metals collect at the base of the SAPS system and in the downstream settling pond.

Figure 3: Cross-section Diagram of a Typical SAPS



Constituents Treated

SAPS treat acidity and can reduce concentrations of aluminum, copper and iron.

Operations

Skousen (2000) describes a SAPS system consisting of acid water ponded from 1 to 3 meters over an organic compost, which is underlain by 0.5 to 1 meter of limestone. The organic matter stimulates the growth of sulfate- and iron-reducing bacteria and enhances or creates reducing

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[16] Skousen, 2000.

[17] EPA, 2004.

conditions, which prevent subsequent armoring of the limestone layer. Drainage pipes below the limestone convey the water into an aerobic pond. The iron and sulfate can be reduced as the water passes through the organic compost.

Data from Demchak et al. suggest that a minimum compost depth of 50 to 60 centimeters is necessary for generating reducing conditions and that the selected compost should decompose at a slow rate.[18]

Optimizing piping system design can improve metal removal. Preferential flow may occur when using only one or two pipes; more pipes allow for more even flow.[19] Precipitated aluminum and iron hydroxides can clog the pore space in the limestone. To assess this effect, the laboratories of the Division of Materials and Minerals at Cardiff University constructed a full-scale SAPS (in cross-section).[20] Study results suggested that the bed permeability will decrease over time, and the SAPS will become clogged. The decreases in permeability are a function of the water flow rates, substrate porosity, potential clogging and influent metal concentrations. Therefore, some sites may require more complex designs that incorporate more treatment cells and settling ponds constructed in series.[21]

Long-Term Maintenance

Regular maintenance is required to prevent system clogging and replenish the compost material. Demchak et al. (2001) suggests adding new compost material after two to three years. Monitoring for overflows and the pressure on the influent side of the system can indicate system clogging and serve as an indicator that the limestone media may need to be replaced. Proper flow control by diligent operation and maintenance efforts is essential to sustained system success.

System Limitations

Longevity is a concern for SAPS, especially in terms of water flow through the limestone. The systems can be prone to clogging without regular maintenance. In addition, the dissolved oxygen concentration of the influent is often a design limitation for SAPS.[22] More complex systems require additional design, construction and management, cost more, and also require a larger footprint.

Costs

The average estimated cost for a SAPS based on a 15-year system life ranges from $72,439 (2013 USD) per year for a 5-gallons per minute (gpm) system to $150,983 (2013 USD) per year

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[18] Demchak et al., 2001.
[19] Demchak et al., 2001.
[20] Rees et al., 2001.
[21] Demchak et al., 2001.
[22] Kepler, 1994.

for a 100-gpm system.[23] For the 5-gpm system, estimated treatment costs are $0.03 per gallon of acid mine drainage. For the 100-gpm system, estimated treatment costs are $0.003 per gallon.

Effectiveness

Long-term metals removal is possible, provided the water chemistry is first fully characterized and the treatment design is based on the geochemistry results.

Table 3: SAPS Removal Efficiencies at Summitville Mine

Metal Removal Efficiencies	
Aluminum	97%
Copper	90%
Iron	64%
Manganese	11%
Zinc	57%
Source: EPA, 2004.	

Kepler and McCleary (1994) verified the success of three SAPS in Pennsylvania. The same researchers (Kepler and McCleary, 1997) reported the use of SAPs in Ohio, Pennsylvania and West Virginia.[24] In all cases, aluminum precipitated in the systems and was flushed using a specialized system (see the Aluminator© technology summary for more information). Table 4 shows the treatment efficiencies observed at five SAPS sites.

Table 4: Removal Efficiencies at Five SAPS Sites

SAPS	Acidity, mg/L as CaCo$_3$		Total Iron (mg/L)		pH		Total Aluminum (mg/L)	
	In	Out	In	Out	In	Out	In	Out
Howe Bridge	320	93	NA	NA	NA	NA	NA	NA
Schnepp Road	84	5	NA	NA	NA	NA	NA	NA
REM	173	88	NA	NA	NA	NA	NA	NA
Buckeye	1,989	1,000	1,005	866	4.0	5.9	41	<1
Greendale	925	150	40	35	2.8	6.5	140	<1

Source: Kepler and McCleary, 1994 and 1997.
NA – Not available

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[23] EPA, 2004.

[24] Kepler, 1994; Skousen, 2000.

Technology: Aluminator©

Technology Description

The Aluminator© is a modified version of Damariscotta's SAPS design. It is an adaptation of a limestone drain in which aluminum hydroxide will accumulate for metals recovery. Instead of using the impervious cap present in limestone drains, the Aluminator© system uses an organic layer and water. The system is designed to operate under high iron and oxygen concentrations, increase pH values and generate alkalinity, retain aluminum in the treatment system, and carry out these functions with a minimum of operation and maintenance requirements.[25] The Aluminator© passive treatment system can treat a wide range of MIW flow rates. This technology does not appear to have been widely applied.

Constituents Treated

The Aluminator© treats aluminum in MIW. It can also treat acidity and remove iron.

System Operations

The Aluminator© is set up similar to a SAPS, in which the base layer increases the pH of the flow, generates alkalinity and retains aluminum. The additional design feature of the Aluminator© allows for regular flushing of the system to remove precipitated aluminum buildup, retaining the system's effectiveness.[26] Treated water is collected in a perforated piping system in the limestone layer and is discharged (Figure 4). These pipes are specifically designed to the total flow and aluminum concentrations. Valves in the piping system can be opened to flush the system using the natural head of pooled water.

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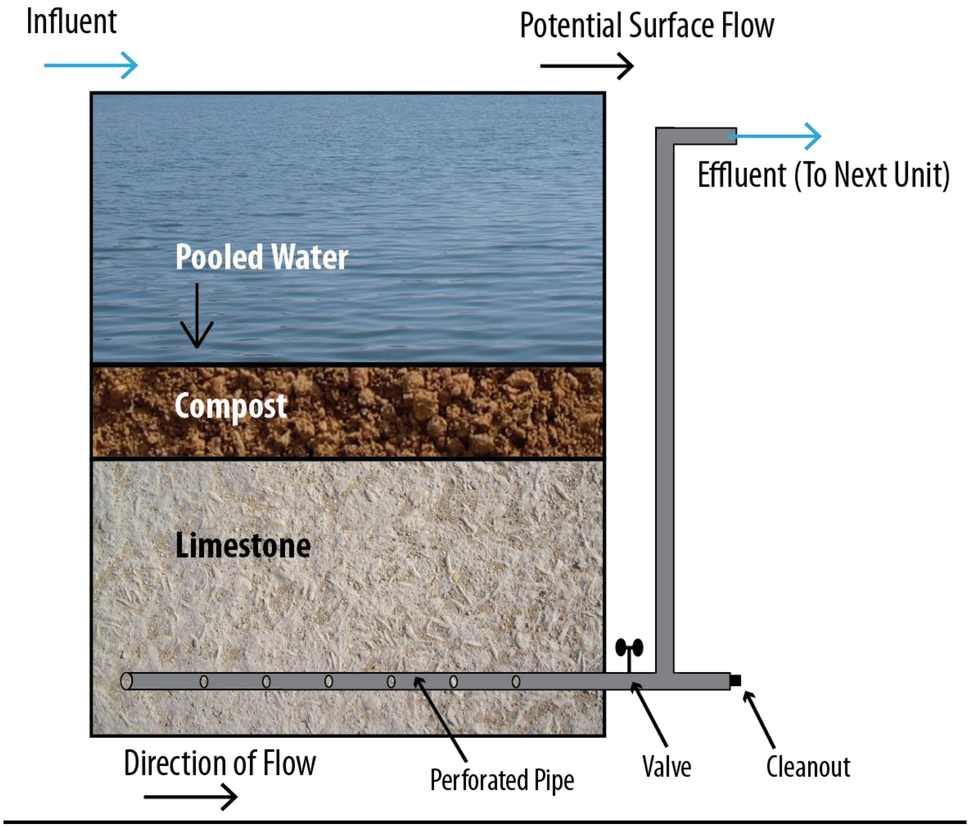
[25] Kepler, 1994.

[26] Kepler, 1994.

Figure 4. Cross-Section of Typical Aluminator© System



Research conducted by D.A. Kepler and E.C. McCleary (1994) included descriptions of Aluminator© usage at the Buckeye Reclamation Landfill in Belmont County, Ohio, and the Little Mill Creek site in Jefferson County, Pennsylvania. The influent and effluent values for the Buckeye and Little Mill Creek sites are included in Table 5. The effluent concentrations reflect the Aluminator© portion of the treatment system. They were achieved immediately after implementation and remained steady.

Table 5: Buckeye Reclamation Landfill and Little Mill Creek System Results

		pH	Alkalinity mg/L $CaCO_3$ equivalent	Acidity mg/L $CaCO_3$ equivalent	Total Iron (mg/L)	Total Aluminum (mg/L)	Total Manganese (mg/L)
Buckeye Reclamation Landfill	Influent	3.98	0	1,989	1,005	41	4.2
	Effluent	5.88	241	1,206	866	0.2	4.0
Little Mill Creek	Influent	2.9	0	325	75	20	20
	Effluent	6.1	54	167	68	<0.1	19

Long-Term Maintenance

Although Aluminator© systems require less operation and maintenance than active treatment systems, some maintenance is still required. System efficiencies decline as interstitial spaces in the substrates fill up, requiring occasional disposal of built-up wastes. Treatment effectiveness can be maintained by periodically removing aluminum from the limestone with minimal interruption of treatment. Greater than 80 percent of accumulated aluminum can be removed from the system with a single flushing of the limestone.[27] A downflow system operating under a positive head is inherently better suited to both accumulating and flushing precipitates than a lateral flow treatment strategy.

System Limitations

At sites with significant levels of aluminum and a low pH, system effectiveness declines more rapidly. Efficiency can also decrease with sustained or uncontrolled high flow events. The technology is limited to the withdrawal of aluminum and iron from MIW.

Costs

The Metro Coal Run watershed in Somerset County, Pennsylvania, consisted of two separate Aluminator© systems, referred to as M1 and M2, each addressing an individual flow from circa 1930 deep mine seals.[28] The M1 Aluminator© was built in spring 2001 and has a bottom footprint of 70 feet by 140 feet. M1 holds roughly 2,900 tons of AASHTO #1 limestone and 240 cubic yards of organic compost, and has a design flow of 85 gpm. The M2 Aluminator© was built in fall 2001 and has a bottom footprint of 40 feet by 200 feet. M2 incorporates 2,200 tons of

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[27] Kepler, 1994.

[28] Kepler, 1993.

AASHTO #1 limestone and 245 cubic yards of organic compost. A typical M2 flow may be 30 gpm; short duration peaks of 200 gpm can be expected. The Metro Aluminator© systems were constructed with roughly 90 percent $CaCO_3$ limestone, thereby requiring the dissolution of less than 300 tons of stone per year for complete neutralization. The total capital cost of the M1 system was about $217,636 (2013 USD). The total capital cost of the M2 system was about $131,901 (2013 USD).

Effectiveness

Results from the Buckeye Reclamation Landfill and Little Mill Creek sites indicate that these systems can remove aluminum from MIW effectively. Typical results do not fluctuate with seasons or temperatures, although efficiency can decrease with sustained or uncontrolled high-flow events. Over the life of a system, decreased treatment efficiency is inevitable. However, well-designed systems can be renovated as necessary to maintain optimum treatment efficiency.

Technology: Constructed Wetlands

Technology Description

Constructed wetlands are built on the land surface using soil or crushed rock/media and wetland plants. Constructed wetlands can be designed as aerobic wetlands, anaerobic horizontal-flow wetlands and vertical-flow ponds (vertical-flow wetlands). Constructed treatment wetlands are designed to treat contaminants over a long period and can be used as the sole technology, where appropriate, or as part of a larger treatment approach. Contaminants are removed through plant uptake, volatilization and biological reduction. The soil- and water-based microbes remove dissolved and suspended metals from acid mine drainage. The primary advantages of wetland treatment are low capital and operation and maintenance costs.

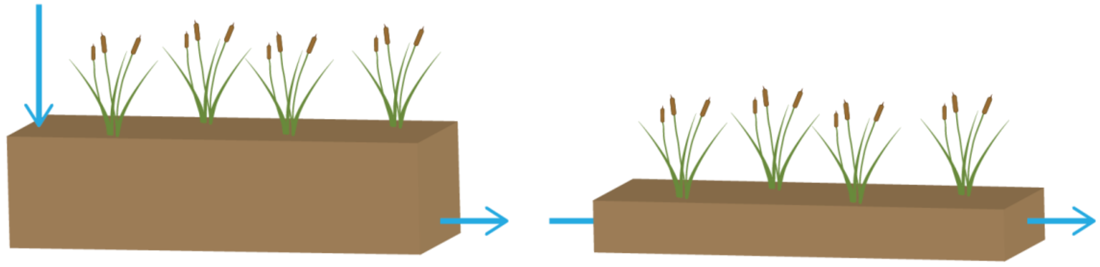
Constituents Treated

Wetlands can capture or treat sulfate and various metals, including iron, manganese, arsenic, aluminum, copper, zinc, cadmium, selenium, nickel and lead. Wetlands can treat acidic, neutral or alkaline mine drainage.

Operations

Constructed wetlands are primarily either subsurface flow wetlands or free water surface wetlands (Figure 5).[29] In a subsurface flow system, water flows through a granular media, such as gravel or sand, and the media surface is planted with aquatic plants. A free water system is similar to a natural wetland or marsh, with water flowing over the surface of a planted treatment cell.

Figure 5. Vertical Flow and Horizontal Flow Constructed Wetlands



The design of a constructed wetland for the treatment of acid mine drainage varies with the site's characteristics. The most important design considerations are biochemical processes, loading rate

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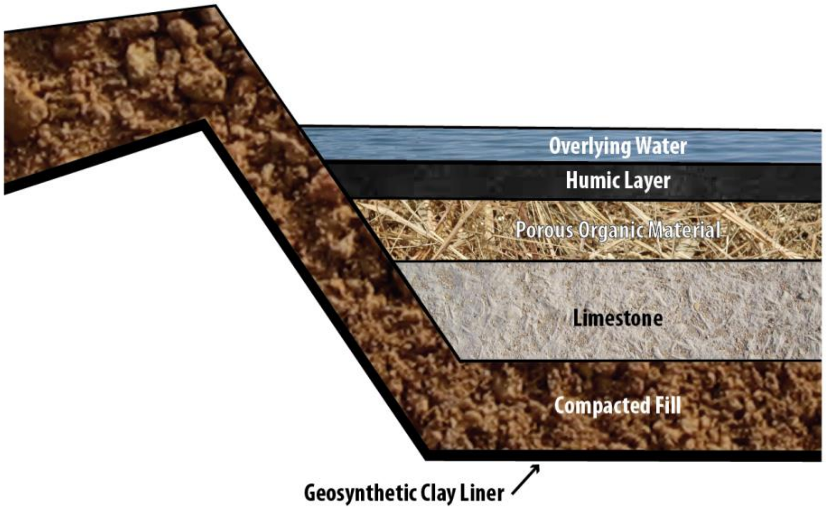
[29] Golder, 2009.

and retention time, slope, substrate, vegetation, sediment control, geometric configuration, seasonality, and regulatory requirements.[30]

If the water is alkaline, aeration to enhance metal oxidation processes can be achieved with an aerobic wetland. Neutralization of acidic water may be required, through the use of an ALD or other pre-treatment. If the water is too acidic or if it contains too much ferric iron, aluminum or dissolved oxygen, a compost wetland could increase alkalinity.

Figure 6. Cross-Section of a Constructed Wetland Design



Long-Term Maintenance

Constructed wetland systems are generally easy to maintain. Monitoring for saturation, spillover and sedimentation is needed. Periodic dredging of sediments may be necessary. Wetlands work well in remote locations or situations where constant monitoring or maintenance may be impractical.

System Limitations

Constructed wetlands have several constraints: 1) they require a large amount of land per unit volume of water; 2) a constant and sufficient supply of water is necessary to support the wetland; 3) influent MIW may require pre-treatment; and 4) periodic release of captured contaminants may occur during high-flow periods or periods when vegetation decomposes, if pH changes and resolubilization occurs or when desorption occurs.[31]

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[30] Rani et al., 2011.
[31] CH2M Hill, 2010.

Seasonality of the locale is an important factor in wetland design, particularly in areas where climate fluctuations are significant. In cold climates, wetland removal efficiencies may decline during winter.[32]

Wetland implementation requires an initial construction cost and upfront monitoring to make sure the system is functioning and stable. Once in place, wetlands are slow to produce significant results in comparison to active treatment technologies.

Costs

The construction cost of an anaerobic wetland in the Copper Basin Mining District in Ducktown, Tennessee, in 1998 was about $1.4 million (2013 USD).[33] This included the initial removal of waste material and construction of the anaerobic cell. In 2003, the cost of adding the two additional aerobic cells to the wetland was about $380,863 (2013 USD). This included the cost of installing the two cells, adding a rock filter and restoring a segment of habitat.

The U.S. Naval Facilities Engineering Service Center's Remediation Technology Online Help Program lists the costs of constructed wetlands treatment at between $0.15 and $1.00 (2013 USD) per gallon of water treated.[34]

Effectiveness

Aerobic wetlands are generally more effective in removing iron than manganese and perform best at low flow rates (Table 6).[35] Seasonal variations in metal removal efficiency have been noted, with lesser amounts removed in cold weather.

Table 6. Treatment Efficiencies Observed for Constructed Wetlands

Constituent	Typical Removal
Acidity	75-90%[1,2]
Sulfate	10-30%[1,2]
Iron	>80-90%[1,2]
Aluminum	>90%[1,2]
Copper	>80-90%[2]
Zinc	>75-90%[2]
Cadmium	>75-90%[2]
Lead	>80-90%[2]

Source: http://www.itrcweb.org/miningwaste-guidance/to_const_treat.htm.
1. For coal mine drainage
2. For metal mine drainage

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[32] Rani, 2011.
[33] FRTR.
[34] ITRC, 2010.
[35] AML, 2004.

After construction of the Copper Basin Mining District initial anaerobic wetland in Tennessee, a performance study found that the wetland successfully reduced the acidity and concentration of most of the metals, with the exception of manganese (Table 7).[36] To help reduce the concentrations of manganese, two additional aerobic cells were added to the wetland system in 2003. In addition, a rock filter was added to provide oxygenation, volatilization of hydrogen sulfide and settling of metal precipitates.

Table 7. Maximum Post-Treatment Concentrations at Copper Basin Mining District Constructed Wetland, 2004 to 2006

	Influent (mg/L)	**Effluent (mg/L)**
pH	4.28	7.16
Aluminum	1.423	0.055
Iron	0.211	0.133
Manganese	1.148	0.294
Copper	0.197	0.017
Zinc	0.640	0.197
Sulfate	110	104
Hardness	97	142
Flow (gpm)	294	241
Acidity	37	<1
Alkalinity	<1	45

Source: http://www.epa.gov/aml/tech/cuwetlands.pdf.
All values are total concentrations, except for manganese, which is dissolved.

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[36] EPA, 2006.

Technology: Biochemical Reactors

Technology Description

Biochemical reactors (BCRs), or bioreactors, use microorganisms to remove contaminants from MIW. BCRs can be constructed in various designs, including in open or buried ponds, in tanks, or in trenches between mine waste and a surface water body.[37] BCRs can address a wide range of flows, acidity and metals loading. They can also operate passively or actively. Passive bioreactors have successfully treated acid mine drainage-contaminated waters in pilot and full-scale projects. Most BCRs at mine sites include sulfate-reducing bacteria and operate anaerobically. As the sulfate is reduced in these systems, metals sulfides are precipitated and removed.

Figure 7. Example of Passive BCR System

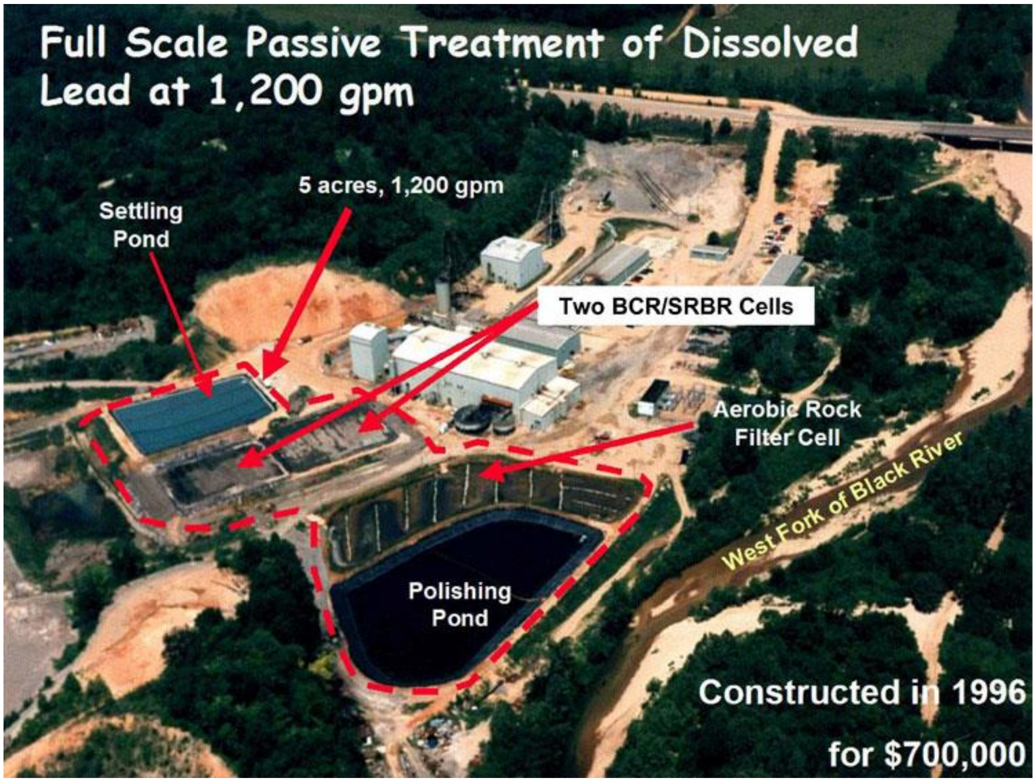

Image Source: http://www.itrcweb.org/miningwaste-guidance/to_bioreactors.htm.

Constituents Treated

BCRs effectively increase pH and remove sulfate and metals such as zinc, lead, copper, cadmium, cobalt and nickel from MIW. Metals and metalloids (e.g., arsenic, chromium, selenium, thallium and uranium) may also be removed.

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[37] ITRC, 2010.

Operations

BCRs can be active or passive systems. Active systems include an external liquid source of organic substrate and separate tanks or zones for the biological, chemical and physical (solids separation) processes.[38] Passive BCRs use a simple flow through design, with a solid reactive mixture acting as a source of carbon for the bacteria and as a substrate for microbial attachment and metal sulfide precipitation. In passive bioreactors, water flows through a solid reactive mixture contained in a pond or a tank that consists of an organic carbon source, a bacterial source or inoculum, a solid porous medium, a nitrogen source and a neutralizing agent.

Passive BCRs can receive the flow from the top, allowing for downflow through the system, from the bottom, with an upflow through the system, or horizontally.[39] The flow design dictates the system transport and removal of metals.

The efficiency of passive bioreactors depends on the activity of bacteria, which is mainly controlled by the composition of the reactive mixture. The organic substrate is generally a mixture of locally available organic materials, and often contains limestone to provide additional neutralizing capacity and gravel to increase substrate permeability. Selection of the organic carbon source is usually based on availability and costs of the added electron donor per unit of reduced sulfate. Sources of organic carbon include liquid ethanol, manure and wood chips, as well as other materials such as fish bones, chitin, biosolids waste fluid and biosolids.[40] BCR applications indicate higher success in systems with a mixture of several contaminants as opposed to a single organic carbon source.[41]

Flow equalization or pH adjustment may be required as a pre-treatment step to generate consistent flow rates and pH to improve the effectiveness of BCRs. Minimum contact times generally range from eight to 48 hours.[42] Following treatment, post-treatment polishing steps oxygenate the effluent and reduce toxicity.[43]

Long-Term Maintenance

In general, BCRs do not require extensive maintenance. However, performance can decline due to substrate clogging. Clogging of the substrate can be reduced by adding inert gravel to the mixture to provide structure to offset compaction. A replaceable cartridge system is an example of such a substrate. Recycling systems in compost-based BCRs and flushing circuits in rock-based BCRs have been used to remove any buildup in the BCR.[44]

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[38] ITRC, 2010.
[39] Zagury, 2007.
[40] ITRC, 2008.
[41] Zagury, 2007.
[42] ITRC, 2010.
[43] Butler et al., 2011.
[44] ITRC, 2010.

System Limitations

The design of a BCR is based on the specific parameters and characteristics of the influent, primarily the pH, flow, temperature, and the type and concentration of metals, as well as the space available for the treatment system.[45] In general, BCRs are built at least 3 to 4 feet deep. Greater depths may be necessary in cold climates. In addition, extended periods of cold can result in reduced performance, and high flows can stress the treatment system, although insulation of the system can minimize the impact of below-freezing temperatures.[46] Impacts on the local area must also be considered. In particular, proximity to populated areas can be problematic due to the strong odors and initial discoloration of the effluent.

Costs

The bioreactor system at the West Fork Mine site in Missouri was built for about $1 million (2013 USD) in 1996. It has a 1,200-gpm capacity.[47] In 2008, construction bids for a similar system at Soudan Underground Mine State Park in Minnesota included a bid at $650,978 (2013 USD).[48] A horizontal system was installed at the Prospector Square Development site in Park City, Utah, in 2008 for approximately $400,000.[49]

A full-scale, compost-free bioreactor system was built at the Leviathan Mine in Alpine County, California, in 2003. Capital costs for construction of the gravity-flow operation were $1,062,100.[50] Changing to the recirculation mode added nearly $38,000, for a total cost of about $1.1 million (2013 USD). Operating at an average flow rate of 10 gpm, the system's operation and maintenance costs are $19.51 (2013 USD) per 1,000 gallons of treated acid mine drainage.

Effectiveness

The Luttrell BCR, located near Helena, Montana, is part of the Tenmile Creek Superfund site. It has treated leachate from an on-site repository since 2003. Data indicate removal efficiency of 95 percent for most metals.[51] Additional studies indicate that systems well suited to site-specific characteristics are able to reduce acidity and remove more than 95 percent of metals.[52] Various types of media for gravel pit seepage can result in 98 percent removal of selenium and achieve less than 5 micrograms per liter of selenium.[53]

In 2007, EPA conducted a pilot study at the Standard Mine in Colorado that demonstrated passive, biological treatment of MIW is feasible at cold, remote, high-altitude sites.[54] Results

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[45] ITRC, 2010.
[46] Reisman, 2008.
[47] Doshi, 2006.
[48] Eger, 2009.
[49] Fitch, 2009.
[50] ITRC, 2006.
[51] ITRC, 2010.
[52] Reisman, 2008 and 2009.
[53] CH2M Hill, 2010.
[54] Reisman, 2009.

from the first 13 months of BCR operation demonstrated high levels of metal removal, with removal rates for cadmium, copper, lead and zinc of about 98 or 99 percent. However, despite the high removal rates, BCR effluent continued to exceed discharge standards for cadmium, lead and zinc.[55] In addition, toxicity assessment of treated effluent from on-site passive bioreactors has yet to take place on a regular basis. Studies at Luttrell indicate that in-field aeration improved the removal of acute and short-term sub-chronic toxicity, most likely through removal of hydrogen sulfide produced within the BCRs.[56]

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[55] Reisman, 2009.

[56] Butler et al., 2011.

Technology: Phytotechnologies

Technology Description

Phytoremediation technologies use plants to treat or capture contaminants in various media. Phytoremediation mechanisms include extraction of contaminants from soil or ground water; hydraulic control of contaminated ground water; and control of runoff, erosion and infiltration by vegetative covers. The mechanisms for removal of contamination by phytotechnologies include concentration of contaminants in plant tissue; degradation of contaminants by various biotic or abiotic processes; volatilization or transpiration of volatile contaminants from plants to the air; and immobilization of contaminants in the root zone.[57] Identifying the appropriate plant species and soil amendments is essential to treatment success. Long-term maintenance is minimal once the vegetation is established.[58]

Constituents Treated

Phytotechnologies can remove metals, including chromium, and radionuclides, including uranium, cesium and strontium. Phytotechnologies are also used to control runoff, erosion and infiltration.

Operations

If time and funding permit, soil or water from a site should be used in lab or greenhouse studies to confirm the effectiveness of the site-specific treatment. Sites with widespread, low-to-medium level contamination in the root zone are the best candidates for phytotechnologies.

Phytoremediation is a sustainable and green technology that does not require supplemental energy.[59] It generates minimal air emissions, water discharge and secondary wastes, and also improves air quality and sequesters greenhouse gases. Phytotechnologies provide restoration and land reclamation benefits during and after cleanup as well as habitat for plants and wildlife.

There are six basic phytoremediation mechanisms that can clean up mine sites: phytosequestration, rhizodegradation, phytohydraulics, phytoextraction, phytodegradation and phytovolatilization (Table 8).

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[57] EPA, 2000.
[58] ITRC, 2010.
[59] ITRC, 2010.

Table 8: Common Phytoremediation Mechanisms Used in Mining Remediation[60]

Mechanism	Description	Cleanup Goal
Phytosequestration	The ability of plants to sequester certain contaminants in the rhizosphere through exudation of phytochemicals and on the root through transport proteins and cellular processes.	Containment
Rhizodegradation	Exuded phytochemicals can enhance microbial biodegradation of contaminants in the rhizosphere.	Remediation by destruction
Phytohydraulics	The ability of plants to capture and evaporate water and take up and transpire water.	Containment by controlling hydrology
Phytoextraction	The ability of plants to take up contaminants in the transpiration stream.	Remediation by removal of plants
Phytodegradation	The ability of plants to take up and break down contaminants in the transpiration stream through internal enzymatic activity and photosynthetic oxidation/reduction.	Remediation by destruction
Phytovolatilization	The ability of plants to take up, translocate and subsequently transpire volatile contaminants in the transpiration stream.	Remediation by removal through plants

Long-Term Maintenance

The majority of maintenance is required during the initial years to ensure the vegetation establishes itself. Replanting or additional soil modifications may be needed, as well as irrigation, weed and pest control.[61]

System Limitations

Root contact is a primary limitation on phytoremediation applicability. Remediation with plants requires that the contaminants be in contact with the root zone of the plants. The plants must be able to extend roots to the contaminants, or the contaminated media must be moved to within range of the plants. This movement can be accomplished with standard agricultural equipment

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[60] ITRC, 2010.

[61] ITRC, 2010.

and practices. However, because these activities can generate fugitive dust and volatile organic compound emissions, potential risks may need to be evaluated.[62]

Pests, infestations and the availability of water as an irrigation source are all possible limitations associated with this technology. However, proper plant selection can overcome many of these limitations. Phytotechnologies also typically require larger tracts of land than many remedial measures.

Phytoremediation is also limited by seasonality and the growth rate of the plants. More time may be required to phytoremediate a site as compared with other, more traditional cleanup technologies. High concentrations of contaminants may also inhibit plant growth and thus limit application on some sites or some parts of sites.

Costs

Total system costs for some phytoremediation applications can be 50 to 80 percent lower than for active treatment technologies.[63] Establishment of phytotechnology systems includes various activities and expenses, such as earthwork, labor, planting stock, planting method, field equipment, heavy machinery (typically farming or forestry equipment), soil amendments, permits, water control infrastructure, utility infrastructure and fencing. According to the ITRC, 10 to 15 percent of a project's initial capital costs should be added as a contingency for replanting.[64] Operation and maintenance costs can include labor, sampling, analytical work, materials, field equipment, utilities, waste handling and disposal. Once the vegetation becomes established, operation and maintenance costs tend to diminish.

Effectiveness

Factors affecting the effectiveness of phytoremediation include the contaminant type and concentration, soil conditions, hydrogeologic conditions, the plant species and growth rate, and climate conditions. Sites with low concentrations of contaminants over large cleanup areas and at shallow depths are best suited for phytoremediation. Long-term monitoring may be needed to demonstrate the effectiveness and to prevent the introduction of contaminants to the human food chain.

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[62] EPA, 2000.

[63] EPA, 2000.

[64] ITRC, 2010.

Technology: Permeable Reactive Barriers

Technology Description

A permeable reactive barrier (PRB) consists of a permeable treatment zone in which reactive material has been placed and through which contaminated water flows (Figure 8). With most PRBs, reactive material is in direct contact with the surrounding aquifer material. Reactive materials include ZVI, limestone, compost, zeolites, activated carbon and apatite.

When properly designed and implemented, PRBs are capable of remediating many different contaminants to regulatory concentration goals. Data indicate that these systems, once installed, will have extremely low, if any, maintenance costs for at least five to 10 years.[65] There should be no operational costs other than routine compliance and performance monitoring.

Figure 8: PRB Diagram

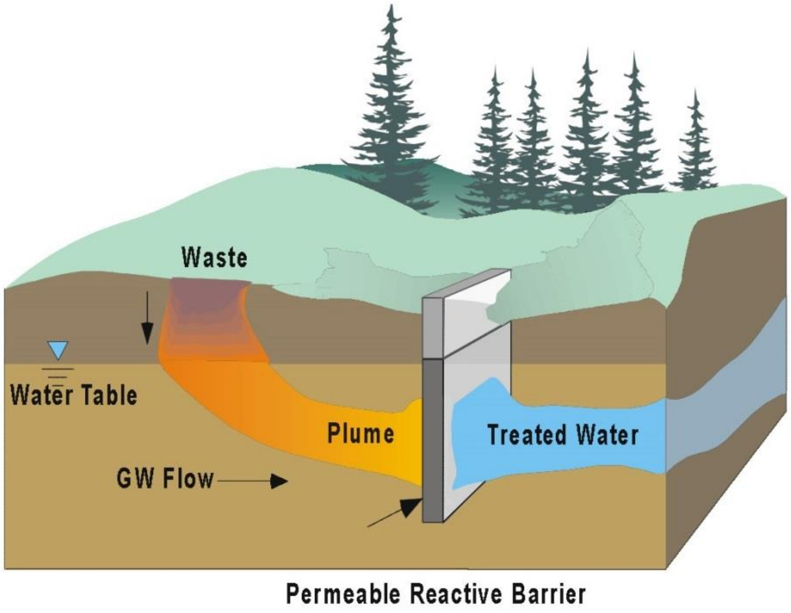

Image Source: http://clu-in.org/download/rtdf/prb/reactbar.pdf.

Constituents Treated

This technology has successfully treated many different constituents, including radionuclides, trace metals, and anion contaminants. Trace metals treated include hexavalent chromium, nickel, lead, uranium, technetium, iron, manganese, selenium, copper, cobalt, cadmium and zinc. Anion contaminants treated include sulfate, nitrate, phosphate and arsenic.

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[65] EPA, 1998.

Operations

PRBs are currently built in two basic configurations: the funnel-and-gate and the continuous PRB. Newer techniques for emplacing reactive media include injection of slurries, hydro-fracturing and driving mandrels. PRBs can be installed as permanent or semi-permanent units. The most commonly used PRB configuration is a continuous trench in which the treatment material is backfilled. The trench is perpendicular to and intersects a ground water plume.

The funnel-and-gate PRB design uses impermeable walls (e.g., sheet pilings, slurry walls) as a "funnel" to direct the contaminant plume to a "gate" containing the reactive media. In contrast, the continuous PRB design completely transects the plume flow path with reactive media. Due to the funnels, the funnel-and-gate design has a greater impact on altering the ground water flow than the continuous PRB. In both designs, it is necessary to install the PRB to the bedrock and to keep the reactive zone permeability equal to or greater than the permeability of the aquifer to avoid diversion of the flowing waters around the reactive zone.

Reactive material is placed in the subsurface where a plume of contaminated ground water passes through, typically under natural pressure conditions, and treated water comes out the other side. The PRB is a barrier to the contaminants, not to the water. The basic objective of any PRB treatment material is to either directly degrade or immobilize target chemicals in ground water or to change the geochemical conditions of the ground water system such that the destruction or immobilization of the target chemicals is promoted.

ZVI is the most widely used reactive material in PRBs, due to its ability to treat common organic and inorganic contaminants in ground water.[66] Other iron- and non-iron-based materials are also being used in pilot-scale and full-scale PRB applications or are being evaluated in laboratory or bench-scale demonstration projects because of their ability to treat additional contaminants, such as radionuclides, heavy metals and MIW.

A complete site characterization is of critical importance for the design and installation of a reactive barrier. The characterization should include an evaluation of the surface features, structures and buried services. This characterization will determine whether the site is amenable to PRB installation and the types of PRB emplacement technologies feasible at the site.

Several important issues must be addressed when considering PRB technology for contaminant remediation. The plume location and extent, ground water flow direction and velocity, and contaminant concentrations must be known to achieve the required performance. In addition, information on stratigraphic variations in permeability, fracturing and aqueous geochemistry is needed for the PRB design. The plume must not pass over, under or around the PRB. The reactive zone must reduce the contaminant to concentration goals without plugging with precipitates.

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[66] ITRC, 2005.

Long-Term Maintenance

Once in place, minimal maintenance is required for PRBs. However, performance can decline over time due to clogging.[67] Precipitates will eventually cause a reduction in permeability; therefore, removal of precipitates could represent a significant operation and maintenance cost for PRB technology. Based on many laboratory evaluations of porosity loss, and fewer evaluations at field sites, some maintenance could be required in highly mineralized and/or oxygenated ground water as frequently as every five years to manage potential problems caused by precipitate formation. In less mineralized waters, maintenance frequency could be as low as every 10 to 15 years.[68]

System Limitations

PRBs are generally not the only remedy for a site. Additionally, since most PRBs operate passively, site remediation may take several years or even decades, requiring the use of long-term institutional controls for site management. Therefore, a PRB should be considered in the context of overall and long-term site remediation goals.

Biofouling and mineral precipitation may limit the permeability of the wall system if not managed properly. Disposal issues could develop in the PRB treatment media after the contaminants are concentrated within the barrier system. This factor is most important in PRB systems that retain the contaminants (e.g., metals and radionuclides), as opposed to PRB systems that degrade the contaminants (e.g., hydrocarbons and chlorinated organics) as they flow through the system.

Costs

The passive functioning of a PRB means that relatively little energy or labor input (except for site monitoring) is necessary; thus, the technology has a potential advantage over conventional ground water treatment methodologies, such as pump-and-treat systems. Overall, while the costs of PRB systems vary depending on site-specific circumstances, the length and especially the depth tend to be the biggest factors that drive the cost of the installation. Once installed, PRBs are generally less expensive to maintain than active systems.[69]

A PRB was installed at a uranium tailing site in Durango, Colorado, at a cost of about $29.68 per 1,000 gallons treated (2013 USD).[70] For a PRB installed in Monkstown, Northern Ireland, total treatment costs were $1.4 million (2013 USD).

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[67] CH2M Hill, 2010.

[68] EPA, 1998.

[69] ITRC, 2010.

[70] US DOE, 2004.

Table 9. PRB Cost Summary, Monkstown Site

Activity	Final Cost (2013 USD)
Site Investigation	
Main site investigation	$371,261
Additional site investigation	$71,381
Subtotal:	$442,642
Remediation	
Soil removal and disposal costs	$144,629
Pilot-scale evaluation	$34,711
Design preparation of contracts and working plan	$30,854
Installation of cutoff wall and PRB	$486,455
Supervision	$73,357
Completion report	$20,268
Subtotal:	$790,274
Ground Water Monitoring	
Monitoring (10 years)	$170,070
Tracer test	$15,427
Consumables	$385
Subtotal:	$185,882
Total:	$1,418,798
Source: Beck, Harris and Sweeny, 2001.	

Effectiveness

As of 2005 in the United States, there were more than 90 applications of iron-based PRBs; 67 of these were full scale.[71] At the uranium tailings site in Durango, Colorado, sampling in 2004 indicated that the influent to the PRB contained 359 µg/L selenium and the effluent contained 8 µg/L selenium.

At the Monticello Mill Tailings site in Utah, the full-scale PRB treatability study using ZVI was put in place in 1999. The PRB had successfully reduced uranium, vanadium, arsenic, selenium, molybdenum and nitrate during the first four years of operation. However, the hydraulic conductivity of the treatment area decreased and caused ground water to mound upgradient of the PRB. Additional investigations identified the need for a supplemental remedy for ground water treatment.

In 2005, a pilot-scale PRB with granular ZVI was installed at the former ASARCO East Helena metal smelting facility, located near Helena, Montana to treat arsenic in ground water. Monitoring results indicate that upgradient arsenic concentrations of greater than 25 mg/L were reduced to concentrations below 2 mg/L.[72] After two years of operation, monitoring indicated significant decreases in arsenic concentrations.

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[71] ITRC, 2005.

[72] EPA, 2008.

Technology: Fluidized Bed Reactor

Technology Description

In a fluidized, or pulsed, bed reactor, contaminated water passes through a granular solid media, such as sand or granular activated carbon, at high enough flow rates to fluidize the media, creating a mixed reactor configuration for attached biological growth or biofilm (Figure 9). Fluidized bed reactors (FBRs) can be designed as aerobic or anaerobic systems. Removal of solids following the biological treatment system is required.

Figure 9: Example FBR System Design

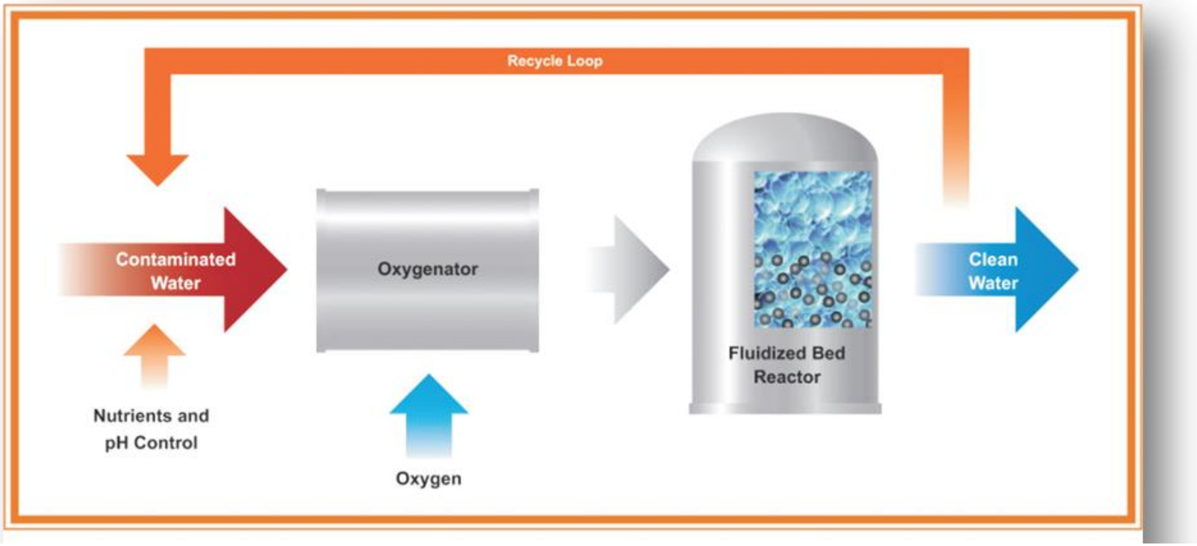

Image Source: http://www.envirogen.com/files/files/FBR_Brochure-draft-Rev_9-4-pges-Sept_2011.pdf.

Constituents Treated

Constituents treated by FBRs include selenium, chromium, nitrate and perchlorate.

System Operations

FBRs are active treatment systems that foster the growth of microorganisms on a hydraulically fluidized bed of fine media. A fluid is passed through granular solid material at velocities sufficient to suspend the solid media. Fluidizing the media increases the active surface area, allowing for increased microorganism growth and contaminant treatment.[73]

During start up, the FBR is seeded with heterotrophic bacteria. Electron donor materials and nutrients are pumped into the anaerobic FBR to promote microbial growth. The process uses

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[73] Envirogen, 2011.

naturally occurring microbes and biodegradable carbon sources to maintain biomass. Contaminated water is then pumped from a feed tank into the FBR in an upward flowing direction to suspend the media. As microorganisms grow on the media, the fluidized bed height expands. Over time, a biofilm develops on the media surface.[74] In anoxic systems, nitrate and selenium reduction occur on this biofilm. Treated water from the FBR system is then discharged to a downstream liquid/solids separation system.

Figure 10: Envirogen's FBR Model EFB-14

Image Source: http://www.envirogen.com/files/files/FBR_Brochure-draft-Rev_9-4-pges-Sept_2011.pdf.

Anoxic FBR systems can treat inorganic contaminants such as nitrate and selenium.[75] The systems routinely achieve discharge limit conditions. Manufacturers can design these systems as modular units for use in remote areas. Another benefit of these types of systems is their relatively small footprint, which ultimately results in lower total installed costs. Depending on influent characteristics, FBRs may not require extensive pre-treatment for suspended solids.

Following treatment, aeration may be required to increase dissolved oxygen and remove biochemical oxygen demand. Adjustment to pH may also be required prior to and following treatment. Other byproducts requiring post-FBR polishing may include dissolved iron, manganese and sulfide.

Long-Term Maintenance

Due to the biological growth inherent in FBRs, daily cleaning of the influent strainer, tank walls, recycle tank and piping are required. Pre-treatment and post-treatment pH adjustment may be

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[74] CH2M Hill, 2010.

[75] CH2M Hill, 2010, Envirogen 2011.

required, as well as period media replacement. Biosolids must be thickened and dewatered prior to landfill disposal.

System Limitations

Common limitations associated with FBRs include the inability to handle high feed nitrate, selenium and suspended solids concentrations.[76] This is due to problems of rapid plugging, nitrogen gas binding, channeling and high backwash requirements. Additionally, the presence of excessive amount of nitrates will require a proportional amount of carbon or energy sources. An external carbon source is needed to meet influent organic content or chemical oxygen needs. This excess carbon source will also generate some additional biomass.

FBRs are temperature sensitive and do not perform as well below 10° C. Water below 10° C would require thermal treatment to heat the waste stream. While FBRs require high-energy inputs, some abandoned mine flows have sufficient pressure to provide enough head for fluidization of the bed.[77] FBRs do lose efficiency in cold weather environments and the generated sludge may be hazardous, requiring proper disposal.

Costs

FBRs are energy intensive, requiring high operating power costs relative to other systems, and the electron donor and nutrient provision can be expensive. Solids removal, which may require membranes, can greatly add to operating and capital costs. Total installed cost for a 1 million gallon per day (gpd) system is estimated at $11.8 million (2013 USD), but can cost significantly more.[78] The annual operation and maintenance cost for the same system is estimated at $3.2 million (2013 USD).

Effectiveness

Envirogen Technologies offers a patented fixed-film FBR biological treatment technology that reportedly demonstrated the ability to achieve less than 5 micrograms per liter (µg/L) selenium over a 10-month period in treating mining leachate at a coal mining site.[79] This system was designed to handle a flow of 2,800 gpm, with a design temperature of 10° C and a typical hydraulic residence time of 26.8 minutes. Table 10 shows results of this trial. There was also a four-month trial at a Canadian coal mine examining Envirogen FBR technology in the presence of high nitrate levels and at colder influent water temperatures. Table 10 shows results of this trial as well. This system was designed to handle a flow of 2,800 gpm, with a design temperature of 4° C and a typical hydraulic residence time of 56.9 minutes.

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[76] CH2M Hill, 2010.
[77] Riefler, 2007.
[78] CH2M Hill, 2010.
[79] Envirogen, 2011.

Table 10: Coal Mining Trial Operating Results at Canadian Coal Mines

Constituent	Feed Concentration	Effluent Concentration
Site 1		
Dissolved Selenium	22 to 27 µg/L	4.7 µg/L maximum
Nitrate – Nitrogen as N	6 to 8 mg/L	
Dissolved Oxygen	11 to 13 mg/L	
pH	7 to 9	
Site 2		
Dissolved Selenium	200 to 500 µg/L	10 µg/L maximum
Nitrate – Nitrogen as N	15 to 40 mg/L	
Dissolved Oxygen	11 to 13 mg/L	
pH	7 to 8	
Source: Envirogen, 2011.		

The EPA Mining Waste Technology Program tested a pulsed limestone bed technology at the Argo Tunnel water treatment facility in Idaho Springs, Colorado.[80] A 60-gpm system was able to increase pH from 3.0 to 7.0, remove greater than 98 percent iron and aluminum, and produce an effluent alkalinity of about 100 mg/L as calcium carbonate. Manganese and zinc in the effluent required additional treatment. Overall, results indicated a decrease in reagent costs and a decreased sludge volume due to the replacement of lime or sodium hydroxide by limestone as the neutralization agent.

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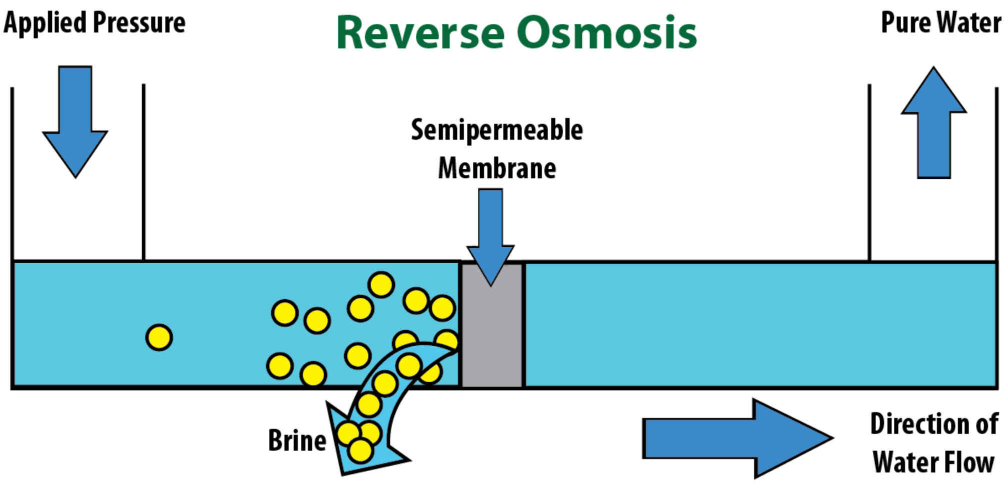
[80] Sibrell, 2005.

Technology: Reverse Osmosis

Technology Description

Reverse osmosis is the pressure-driven separation of contaminants through a semi-permeable membrane that allows water to pass through while retaining contaminants. The dissolved ions are retained in a concentrated brine solution that requires management or disposal (Figure 11). Reverse osmosis is a proven method to demineralize acid mine drainage. However, it does require significant construction and operating costs.

Figure 11: Diagram of Simplified Reverse Osmosis Technology



Constituents Treated

Reverse osmosis can be used to remove metals, total dissolved solids, and sulfate from MIW.

Operations

Reverse osmosis involves a semipermeable membrane through which almost pure water is removed from a concentrated input solution by applying pressure, leaving a more highly concentrated brine solution. The membrane is the primary component of a reverse osmosis system. Durable membranes resistant to chemical and microbial agents that retain structural stability over long operating periods are essential. Pre-treatment is necessary to prevent membrane fouling, particularly if the water contains elevated levels of hardness (i.e., calcium or magnesium) or total suspended solids. With pre-treatment and routine maintenance, membranes typically last two to five years.[81]

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[81] CH2M Hill, 2010.

The brine produced is typically 20 to 30 percent of the influent flow in a single system, depending on influent water quality. The process produces a concentrated waste stream that must be disposed of properly. Various disposal options are available. However, the concentrated waste stream is typically disposed of through evaporation, deep well injection or ocean discharge.

Reverse osmosis has been used successfully to treat MIW at several sites. At the Barrick Richmond Hill Mine near Lead, South Dakota, reverse osmosis polishes selenium from mine water after pre-treatment for iron reduction and precipitation. The reverse osmosis unit is operated at pressures of 250 pounds per square inch and greater.[82] As of 2005, filtration pre-treatment was required to remove total suspended solids. During winter months, water is heated to prevent crystallization caused by depressed salt solubilities. Additionally, a softening plant is under consideration for treating gypsum scaling resulting from elevated calcium concentrations.

At a historic former gold mine in California, reverse osmosis treated impounded water as an emergency measure to prevent impounded water from affecting a drinking water reservoir below the mine.[83] Trailer-mounted reverse osmosis systems were leased along with pre-filtration and manganese removal columns. The system flow was greater than 100 gpm and the system operated for about four months. The system operated at about 40 percent selenium recovery due to the high total dissolved solids in the influent water. The brine was then returned to the initial impoundment. Selenium concentrations were reduced from about 60 µg/L to less than 5 µg/L. The system operated until levels in the impoundment were reduced to acceptable levels.

Reverse osmosis is also in use at the Kennecott South site, which is located in the Salt Lake Valley, east-southeast of Copperton, Utah.[84] The Bingham Canyon Water Treatment Plant (BCWTP), built as part of the site's remedy, is located in operable unit (OU) 2. Reverse osmosis is being used as the primary technology for addressing total dissolved solids- and sulfate-impacted ground water.

The BCWTP has two reverse osmosis treatment racks that treat 3,200 gpm with total dissolved solids concentrations of about 2,000 mg/L and a sulfate concentration of 1,200 mg/L.[85] The quality of the feed water and regular cleaning has extended the system's lifespan to about six years.

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[82] Microbial Technologies, 2005.

[83] Golder, 2009.

[84] ITRC, 2010.

[85] ITRC, 2010.

Figure 12: Two Reverse Osmosis Treatment Racks at the BCWTP

Image Source: http://www.itrcweb.org/miningwaste-guidance/cs48_kennecott_south.htm.

Long-Term Maintenance

Frequent membrane monitoring and maintenance are required to ensure the effective operation of a reverse osmosis system. With pre-treatment and the performance of routine maintenance, membranes typically last two to five years. If membranes become fouled due to lack of pre-treatment or water chemistry, or when they reach the end of their effective lifespan, proper waste disposal is required. Following the performance of a waste determination, proper waste disposal is required consistent with a waste determination.

To minimize viscosity effects that inhibit performance, water temperature controls may be needed at low and high temperatures. Adjustment to pH and the addition of antiscalant to prevent membrane fouling may also be required.[86] Brine concentrates generated during the reverse osmosis process must either be treated or further concentrated for proper disposal. The treated stream may also require pH and total dissolved solids buffering prior to discharge to receiving waters to meet regulatory requirements.

System Limitations

The primary constraint is the need for high operating pressure and associated energy and costs to push the water through the membrane to remove high levels of total dissolved solids. The technology is not practical above 10,000 mg/L total dissolved solids.[87]

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[86] CH2M Hill, 2010.

[87] CH2M Hill, 2010.

Pre-treatment and chemical additions are required to reduce membrane scaling and fouling. Scale-forming ions will irreversibly foul the membranes and create selenium removal issues by allowing leakage. Disposal of brine solution is also required. In addition, low ionic-strength effluent lowers hardness and alkalinity in receiving streams, which can increase susceptibility of aquatic biota to metals toxicity.

Costs

Reverse osmosis typically requires high capital costs for the purchase, installation and operation of the membrane system. For a 1-million-gpd system, the total installed cost is estimated at $42.9 million (2013 USD).[88] Annual operation and maintenance costs for the same system are estimated at $3.2 million (2013 USD). Management and disposal of the brine solution that is generated can require higher operating costs. In arid climates, atmospheric evaporation may offer a technique for removing water in the brine solution followed by appropriate solid waste disposal. For locations where atmospheric evaporation is not feasible, thermal treatment may be needed.

The total cost for the BCWTP was about $16.1 million (2013 USD).[89] Total yearly operation and maintenance costs (40 percent of these costs represent labor and 24-hour maintenance) for the BCWTP are about $1.3 million (2013 USD). These capital and yearly operation and maintenance costs include energy requirements, but do not reflect the costs associated with extraction wells, feed pipelines, disposal infrastructure and off-site disposal.

According to Golder Associates, the capacity of a permanent reverse osmosis facility used to treat waste rock leachate was 60 gpm and included an evaporator for brine management. The capital cost of the system was estimated at $5.6 million (2013 USD).[90] The operation and maintenance cost was estimated at $19.28 (2013 USD) per 1,000 gallons of water treated.

Effectiveness

The BCWTP at the Kennecott South site has consistently seen permeate reduction efficiencies of 71 to 72 percent.[91] Between June 2006 and May 2009, total dissolved solids removal efficiencies at the BCWTP averaged 98.9 percent. Between 2007 and 2009, product water also consistently complied with all applicable State of Utah primary and secondary drinking water standards. Product water continues to remain in compliance with permit limitations established by the State of Utah's Division of Drinking Water.

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[88] CH2M Hill, 2010.

[89] ITRC, 2010.

[90] Golder, 2009.

[91] ITRC, 2010.

At the Barrick Richmond Hill Mine, reverse osmosis concentrates selenium from mine water after treatment by iron reduction and precipitation. Selenium is reduced from 22 µg/L to 12 µg/L to about 2 µg/L at flows of 200 gpm.[92]

Reverse osmosis systems have been demonstrated at full scale to remove selenium (selenate or selenite) to less than 5 µg/L and to remove 90 to 98 percent of total dissolved solids.[93] At a historic former gold mine in California, reverse osmosis was used to treat impounded water for reduction of selenium from about 60 µg/L to less than 5 µg/L, with a flow rate of greater than 100 gpm.

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[92] Golder, 2009.

[93] Golder, 2009.

Technology: Zero Valent Iron

Technology Description

Zero valent iron (ZVI) can be used in active MIW treatment systems to rapidly neutralize acid and promote removal and immobilization of dissolved heavy metals. Adsorption onto the iron metal surface, or onto iron corrosion products initially present on the unreacted metal surface, facilitates the rapid removal of metals from MIW. ZVI can reduce selenium oxyanions to elemental selenium. Ferrous cations can also reduce selenate to selenite and subsequently remove selenite by adsorption to iron hydroxides. In an aqueous environment, ZVI can be oxidized to dissolved ferric (Fe^{3+}) and ferrous (Fe^{2+}) ions. These ions react with hydroxyl ions present in water to form ferric and ferrous hydroxides. Selenate is reduced to selenite while ferrous iron is oxidized to ferric iron. Selenite then adsorbs to the ferric and ferrous hydroxide surfaces and is removed from solution.

Constituents Treated

Constituents treated by ZVI include selenium, arsenic and radionuclides.

Operations

In the treatment of MIW, ZVI is added as a reducing agent and acts as both a catalyst and an electron donor for the reaction.[94] As the ZVI is oxidized, green rust forms, which is required to abiotically reduce selenate to selenite and selenite to elemental selenium.

Anoxic conditions are needed for successful treatment. ZVI treatment systems have typically applied the media in covered tanks or filter vessels that hold elemental iron (Figure 13).[95] Multiple tanks in series can be used to increase treatment. Aeration and oxidation of the iron is needed downstream of ZVI tanks prior to discharging to the outfall.

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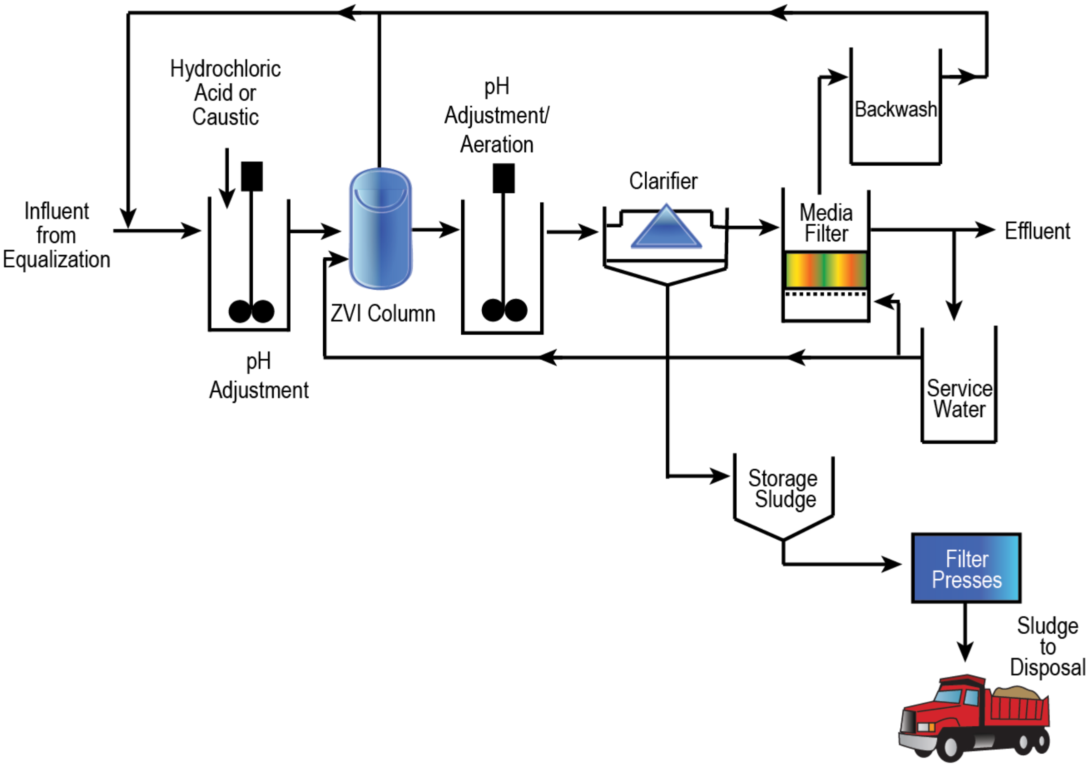
[94] CH2M Hill, 2010.

[95] CH2M Hill, 2010.

Figure 13. ZVI Process Diagram Using Column Setup with Steel Wool



The iron can come in several forms, including filings, steel wool, iron-impregnated foam, and granular micro- or nano-scale particles.[96] The form determines the surface area available for the reactions. The presence of sulfate-reducing bacteria in ZVI systems will offer additional pathways for metal removal (i.e., precipitation of insoluble metal sulfides). Depending on input pH, significant accumulations of precipitates containing iron, aluminum and manganese might be expected.

A 2003 lab evaluation examined the applicability and limitations of granular ZVI for the treatment of water affected by mine wastes.[97] The results indicated metal removal from solution and acid-neutralization occurred simultaneously and most rapidly during the initial 24 hours of reaction. Results indicated that metal removal is most effective in solutions highly undersaturated with respect to pure-metal hydroxides, suggesting that adsorption is the initial and most rapid metal uptake mechanism. Reversibility studies indicate that ZVI will retain metals after shifts in redox states are imposed. However, remobilization of metals may occur after the acid-neutralization capacity of the material is exhausted. Rates of acid-neutralization and of metal (copper, cadmium, nickel, zinc, mercury, aluminum and manganese) and metalloid (arsenic) uptake were determined in batch systems using simulated mine drainage (initial pH 2.3 to 4.5; total dissolved solids 14,000 mg/L to 16,000 mg/L).

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[96] CH2M Hill, 2010.
[97] Wilkins, 2003.

At the Barrick Richmond Hill Mine near Lead, South Dakota, ZVI reduces selenate to selenite, followed by ferric sulfate precipitation at pH 4.5.[98] The process is able to remove selenium to concentrations of 12 to 22 µg/L. At another mine site, ZVI and ferrous sulfate at pH 4.5 are used to reduce selenate to selenite. Ferric chloride is then added to form ferric hydroxide, which co-precipitates the selenite.

In 2008, at the Catenary Post site in West Virginia, steel wool (fibrous ZVI) was used to remove selenium from MIW in a continuous flow reactor.[99] Problems with the treatment process included the formation of iron oxides, passivation of the iron, exhaustion of the iron, and cost.

Long-Term Maintenance

ZVI requires removal, appropriate disposal and replacement, generally within a few months to a few years. Iron residuals with adsorbed selenium typically go through thickening and dewatering processes prior to disposal in a landfill. However, prior to disposal, residuals may require toxicity characteristic leaching procedure (TCLP) testing to determine if the sludge needs to be disposed of as hazardous waste.

System Limitations

Although ZVI can effectively treat MIW, additional treatment methods are likely needed to meet discharge requirements. Pre-treatment may be necessary to adjust pH. Influent temperatures may need to be adjusted to improve removal efficiencies. The formation of green rust begins at a pH greater than 4 and begins to diminish at a pH greater than 5; therefore, a pH range of 4 to 5 is ideal for efficient treatment.[100] The temperature will affect the reaction kinetics, or rate at which selenium reduction occurs. Low temperature will reduce reaction kinetics; therefore, the systems should be designed appropriately. Treatment generally requires a residence time of four hours or greater to adequately remove metals. Passivation of the iron may also occur, depending on the presence of other metal redox reactions, chemical scale and/or suspended solids that will accumulate on the surface of the iron and reduce the reduction capacity. Therefore, over time, the iron will need to be replaced. In addition, the process creates iron oxides and sludge that will need to be removed and disposed of.

Costs

For a column-based system using steel wool, the total installed cost for a 1 million gallons per day (MGD) system is estimated at $13.9 million (2013 USD).[101] The annual operation and maintenance cost for the same system is estimated at $3.3 million (2013 USD). For a stirred-tank based system using granular ZVI, the total installed cost for a 1 MGD system is estimated at $11.8 million (2013 USD). The annual operation and maintenance cost for the same system is

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[98] Golder, 2009.
[99] Golder, 2009.
[100] CH2M Hill, 2010.
[101] CH2M Hill, 2010.

estimated at $3.2 million (2013 USD). There may be additional disposal costs if system wastes are hazardous wastes.

Effectiveness

A 2003 lab study evaluated the applicability of two types of ZVI filings, Fisher iron and Peerless iron, in treating water impacted by acid mine drainage.[102] The pH of all three solutions increased most rapidly within the first 24 hours of the reaction, and increased from initial values of 2.3 to 4.5 to final values of 5.5 to 10.0. These final pH values were controlled by initial pH and the mass concentration of iron. Lower initial pH conditions resulted in final pH values of 5.7 to 6.0, while higher initial pH conditions resulted in final pH values of 9.6 to 10.0. Fisher iron showed similar trends in pH and metal removal efficiency as compared to Peerless iron. Experiments at an initial pH of 2.3 resulted in significant decreases in metal concentrations, except for iron and manganese, which both increased in concentration.

The initial presence of aluminum (III) or iron (III) had little effect on metal removal efficiency. Experiments at an initial pH of 3.5 resulted in decreases of aluminum, mercury, cadmium and nickel to below detection limits, while zinc, arsenic, copper and manganese concentrations were reduced to low levels. Concentrations of iron increased over the first 20 hours of reaction before decreasing to final values below 0.1 mg/L. Experiments at an initial pH of 4.5 resulted in decreases of mercury, cadmium, copper and nickel to below detection limits, while arsenic and manganese concentrations were reduced to low levels. Concentrations of iron increased over the first 75 hours of reaction before decreasing to a final concentration of 0.118 mg/L. Sulfate concentrations were reduced in all experiments; sulfate reduction in solutions with the Fisher iron was greater than reduction in solutions with the Peerless iron.

While iron reduction can significantly decrease selenium concentrations at low selenium concentrations, it has not been proven in full-scale treatment or at high selenium concentrations. At the Barrick Richmond Hill Mine, the process is only able to remove selenium to concentrations of 12 µg/L to 22 µg/L, and additional steps are used to achieve compliance with effluent regulatory limits.[103] In 2008, steel wool (fibrous ZVI) was used to remove selenium from MIW at the Catenary Post site in a continuous flow reactor, with influent concentrations of selenium ranging from 5 µg/L to 14 µg/L. Over 250 days, effluent concentrations of selenium did not consistently achieve the regulatory limit of 5 µg/L. Results indicated a minimum five-hour contact time is required for selenium levels to meet regulatory requirements. At a separate location, a ZVI system piloted at a phosphate mine for removal of selenium was unable to achieve a concentration of 5 mg/L.

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[102] Wilkins, 2003.

[103] Golder, 2009.

Technology: Rotating Cylinder Treatment Systems

Technology Description

The Rotating Cylinder Treatment System™ (RCTS) is an innovative form of lime precipitation treatment. Lime is used to increase the pH of the contaminated water, allowing for the oxidation or precipitation of metals.[104] In typical lime treatment systems, scaling of precipitated metal hydroxides and oxides can reduce or eliminate the efficiency of the system. In an RCTS, water being treated flows through shallow troughs containing rotating perforated cylinders that transfer oxygen and agitate the water, intending to reduce or prevent scaling (Figure 14). RCTS systems can treat highly acidic waters, high sulfate and metals in cold weather or remote locations. RCTS require a post-treatment solids separation unit, such as a pond.

Figure 14: RCTS Unit at the Soudan Mine

Image Source: http://www.asmr.us/Publications/Conference%20Proceedings/2010/papers/0248-Eger-MN-1.pdf.

Constituents Treated

RCTS systems precipitate metals, including aluminum, cadmium, copper, iron, and zinc, and increase pH.

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[104] ITRC, 2010.

Operations

Ionic Water Technologies, Inc. originally developed the RCTS to treat concentrated acid mine drainage containing ferrous iron concentrations of 4,000 mg/L to 7,000 mg/L at the Rio Tinto Mine, which is located near Mountain City, in Elko County, Nevada.[105] It has also been used at the Elizabeth Mine in Vermont and the Gladstone Mine in Colorado.

According to Ionic Water Technologies, the RCTS introduces water to air in a thin film clinging to the rotating perforated cylinder, rather than injecting air into water. When the perforations impact the water, it is aggressively agitated and bubbles are forced into the water. This system of aeration replicates natural aeration and eliminates the need for compressors and blowers. These systems are portable and can be sized to suit the oxidation requirements of individual sites.

This technology has been tested at a site in Alpine County, California.[106] At that test site, the two sources of acid mine drainage emanate from a waste rock pile. Lime neutralization using RCTS for lime mixing and aeration/oxidation of reduced metals was used at this site, operating seasonally during 2004, 2007 and 2008. At this location, two RCTS-60HS systems were required due to the high concentrations of reduced iron in the MIW. The RCTS used less lime to treat comparable flows when compared to a conventional lime precipitation system treating the same water.

Long-Term Maintenance

RCTS systems require minimal operation and maintenance activities. When compared with conventional systems, RCTS systems are designed to require less power and less space, be more effective at mixing, and require less maintenance associated with scaling. However, there can be problems with plugging.

System Limitations

The primary limitation of RCTS treatment involves the sufficient removal of suspended solids. At the Soudan Mine in Minnesota, the addition of an RCTS was successful in raising pH and removing filterable metals.[107] The system of settling tanks and bag and cartridge filters generally reduced total suspended solids to around 30 mg/L, but suspended metals exceeded permit levels. Suspended copper was particularly difficult to remove; even after 48 hours of settling, suspended copper still exceeded permit levels. Given the problems with suspended metals removal and an approaching enforcement deadline, a mobile modular ion exchange system was installed to treat the entire mine discharge.

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[105] Tsukamoto, 2006.
[106] ITRC, 2010.
[107] Eger, 2010.

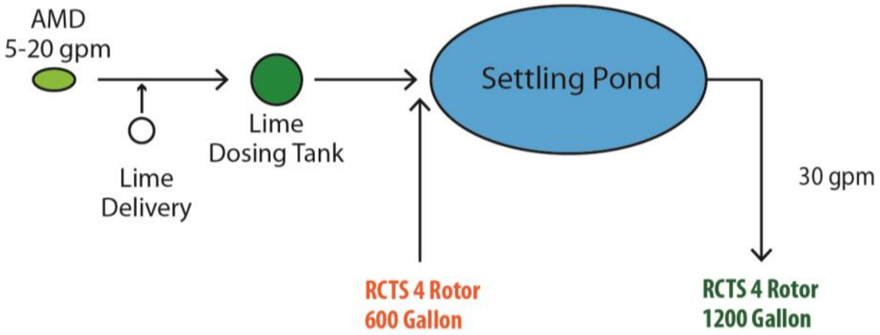
Costs

One RCTS site in California reported a total capital cost of $214,253 (2013 USD) and operation and maintenance costs of $535,633 (2013 USD) for six months.[108] This system can apply most neutralizing chemicals commonly used for mine water treatment. At the Soudan Mine, the estimated installation cost for a peak flow from the mine of 300 gpm was about $135,195 (2013 USD), with an annual operation and maintenance cost of $22,532 to $56,331 (2013 USD), depending on the chemical selection and dose.[109]

Effectiveness

In 2004 and 2005, two four-rotor RCTS units were installed in series to treat acid mine drainage containing more than 7 grams per liter (g/L) of dissolved metals (Table 11) at the Rio Tinto Mine in Nevada (Figure 15).[110] Lime slurry was mixed with impacted water in a lime-dosing tank to raise the pH from about 2.6 to about 8.5. The water from the flash reactor tank was then mixed with RCTS effluent prior to delivery to a settling pond. The pH-adjusted mixture from the settling pond was then fed into the RCTS system. The flow rate to the system ranged from 5 to 20 gpm during two treatment events. The RCTS treatment system effectively removed metals to below applicable water standards at the site. Dissolved metals concentrations were reduced to less than 0.2 mg/L, with the exception of manganese, which was reduced from up to 96 mg/L to less than 0.58 mg/L.

Figure 15: Schematic of the 2005 Rio Tinto Water Treatment, with Two RCTS Units



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[108] ITRC, 2010.
[109] Eger, 2009.
[110] Tsukamoto, 2006.

Table 11: RCTS Treatment Results for Dissolved Metals at the Rio Tinto Mine, 2005

	Influent	**Effluent**	**Influent**	**Effluent**	**Influent**	**Effluent**
Date	*7/26/2005*	*7/26/2005*	*8/5/2005*	*8/5/2005*	*9/28/2005*	*9/28/2005*
Aluminum	793	0.1	540	0.08	325	0.07
Arsenic	0.03	Not detected (ND)	ND	ND	ND	ND
Cadmium	0.359	0.0005	0.338	0.0002	0.198	0.0002
Copper	314	0.005	228	0.002	139	0.009
Iron	6,890	ND	4,990	0.05	2,940	ND
Manganese	96	0.52	80.3	0.41	58.2	0.58
Zinc	79.4	ND	60	ND	36.5	ND
Sulfate	24,180	2,410	17,600	1,800	9,710	2,390

Source: Tsukamoto, 2006.
Concentrations in mg/L

The RCTS has been shown to be effective at treating acid mine drainage, sulfate, and neutral or alkaline drainage. A 2008 demonstration conducted near Gladstone, Colorado, suggested that the system is capable of meeting the applicable water quality criteria for the constituents of concern.[111] The effectiveness of RCTS aeration allows for precipitation of iron and manganese at a lower pH than conventional treatment systems. The RCTS also provides greater oxygen addition per energy consumed than conventional systems. These systems have also proved their ability to successfully produce effluent capable of meeting all federal water quality standards.

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[111] Smart et al., 2009.

Figure 16: RCTS at the Rio Tinto Mine

Image Source: http://www.iwtechnologies.com/pdfs/High_Efficiency_Treatment_Acid_Mine_Drainage.pdf.

At the Soudan Mine, chemical treatment with an RCTS was chosen for additional field evaluation, after bench testing showed that treatment with magnesium hydroxide could reduce copper and cobalt concentrations to permit levels.[112] The RCTS successfully raised pH and removed the metals. However, total suspended solids remained high and exceeded permit limits.

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[112] Eger, 2010.

Technology: Ferrihydrite Adsorption

Technology Description

Ferrihydrite adsorption, also known as iron co-precipitation, is a two-step physical adsorption process that can remove heavy metals from MIW. In low-iron-containing waters, iron may be added to co-precipitate or adsorb certain metals onto ferric hydroxide precipitates. The process involves the addition of a ferric salt to the water to generate a ferric hydroxide and ferrihydrite precipitate, the formation of which results in concurrent adsorption of metals on the surface. The precipitated iron can then be removed. EPA designated the technology a Best Demonstrated Available Technology for selenium removal. It is widely implemented at full scale throughout the mining industry.

Constituents Treated

Constituents commonly treated by this technology include selenium and arsenic and metals that can co-precipitate with iron.

Operations

Ferrihydrite adsorption is a two-step process in which a ferric salt is added to MIW to form ferric hydroxide (Figure 17).[113] Oxidation of ferrous iron to ferric iron and formation of ferric hydroxide is a common reaction in water that causes iron staining because the ferric hydroxide is insoluble and readily precipitates from water. Co-precipitation occurs when some other compound is precipitated along with ferric hydroxide from water. Gravity sedimentation may be required to separate iron solids and adsorbed metals from the water matrix. Generated sludge requires off-site disposal.

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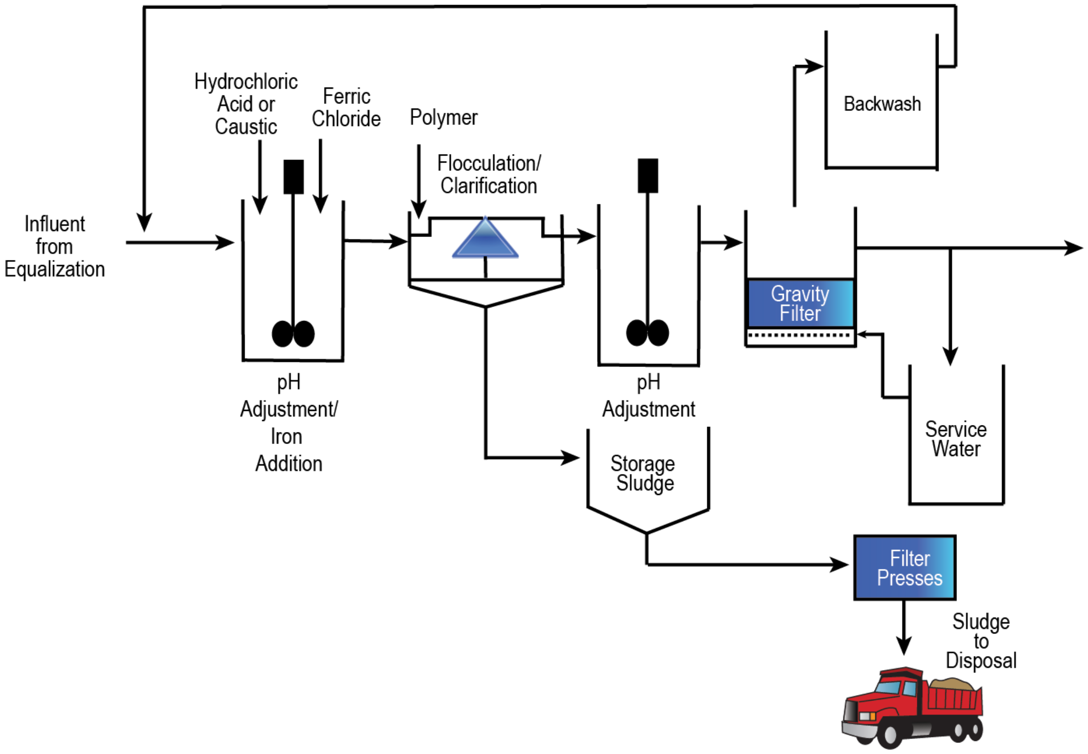
[113] CH2M Hill, 2010.

Figure 17. Ferrihydrite Adsorption Flow Diagram



The ferrihydrite adsorption process is used in various industries for metals removal. It has been shown to be effective for removal of selenium if present as selenite when formed at a pH of 4 to 6.[114] Lime can be used instead of caustic and pressure filtration can be used rather than gravity.

EPA conducted a pilot study of the technology at the Kennecott Utah Copper Corporation's Garfield Wetlands-Kessler Springs site.[115] Results indicated that the technology was effective at reducing selenium concentrations, but that it would not be feasible to use at a large scale at the site due to high costs. However, the technology has not consistently demonstrated that it can achieve regulatory discharge levels of selenium (5 µg/L).

Long-Term Maintenance

Management, dewatering and disposal, typically off site, of the generated sludge is required.[116] Sludge management can include centrifuge, belt press, or plate and frame press. TCLP testing is required to determine whether the sludge is a hazardous waste. At the Garfield Wetlands-Kessler Springs site, TCLP analysis on samples of filter cake indicated it was hazardous waste.[117]

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[114] CH2M Hill, 2010.

[115] MSE, 2001.

[116] CH2M Hill, 2010.

[117] MSE, 2001.

System Limitations

Potential limitations of ferrihydrite adsorption technology include its inability to attain regulatory levels for all metals, such as selenium, the need to manage and dispose of sludges, and high operation and maintenance costs.[118]

Costs

There are high operational costs related to the use of the ferrihydrite adsorption, which is typical of chemical treatment. Costs can increase greatly if residual sludge requires disposal as hazardous waste. Costs reported in literature indicate that the total installed cost of a 1 MGD treatment system is estimated at $11.8 million (2013 USD), with an estimated annual operation and maintenance cost of about $4.3 million (2013 USD).[119]

Effectiveness

Ferrihydrite precipitation with concurrent adsorption of selenium on the ferrihydrite surface is an EPA Best Demonstrated Available Technology. This technology has been widely implemented at full scale and its effectiveness in treating selenium, arsenic and various metals has been demonstrated, proving most effective when used to treat arsenic. A 2000 study suggested the best pH for selenite adsorption ranges between 4 and 6, producing 85 to 95 percent removal of selenite, with decreases in adsorption noted at pHs greater than 7.[120]

In 2001, EPA's 1 gpm pilot demonstration at the Garfield Wetlands-Kessler Springs site demonstrated the effectiveness of the treatment technique. The influent contained 1,950 µg/L of selenium (1,870 µg/L selenate and 49 µg/L selenite). Using an iron concentration of 4,800 mg/L, the mean effluent selenium concentration was 90 µg/L.[121] The minimum reported selenium concentration was 35 µg/L.

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[118] Golder, 2009.

[119] CH2M Hill, 2010.

[120] CH2M Hill, 2010.

[121] MSE, 2001.

Technology: Electrocoagulation

Technology Description

Electrocoagulation involves the application of an electrical current to coagulate organic constituents and suspended solids in water. The affected water is treated using electrolysis with graphite or stainless steel cathodes in conjunction with a metal anode.[122] When a voltage is applied, metals precipitate out of the water. A secondary treatment step such as sedimentation or filtration can then remove the precipitated metals from the water. Mining operators use electrocoagulation to remove suspended particles of clay and coal fines from mine process water. However, electrocoagulation is not a proven technology for full-scale treatment of mining wastes.

Constituents Treated

Constituents commonly treated by electrocoagulation include metals such as arsenic, copper, lead, zinc and cadmium, as well as phosphates and total suspended solids.

Operations

Electrocoagulation is the process of destabilizing suspended, emulsified or dissolved contaminants in an aqueous medium by introducing an electrical current into the medium.[123] The electrical current provides the electromotive force to drive the chemical reactions. Potential mining applications of this technology include final treatment and polishing of discharge water from a high-density sludge water treatment plant, pretreating water prior to filtration or reverse osmosis, and treating neutral tailings water to remove minor amounts of metals prior to discharge.[124]

A typical electrocoagulation unit consists of a chamber with a series of iron or aluminum metal plates.[125] As wastewater flows through the chamber, a direct current is applied to the chamber by attaching positive and negative leads to the first and last terminals. The metal plates are electrified, acting as an induced electrode and releasing metal ions into solution. Precipitates formed by the process can be separated by gravity sedimentation or membrane filtration at higher concentrations. A media filter could be required for tertiary treatment of the gravity sedimentation effluent. Contaminant removal depends on the composition of the water, the material and configuration of the electrodes, and the electric current.

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[122] Golder, 2009.
[123] CH2M Hill, 2010.
[124] ITRC, 2010.
[125] CH2M Hill, 2010.

Electrocoagulation is effective when the influent water has a high electrical conductivity.[126] The treatment has also been demonstrated to work more efficiently when lower concentrations of pollutants are present and when the pH is between 4 and 8.

Electrocoagulation can reduce sludge production significantly compared to other chemical processes such as iron reduction.[127] However, thickening, dewatering and leachate testing must be considered when thinking about residuals disposal. If wastes are determined to be hazardous, waste disposal costs may increase significantly.

Long-Term Maintenance

Overall, electrocoagulation is considered a low-maintenance technology and requires minimal operator attention. Cathode plates in system units require frequent cleaning. Electrodes also need to be replaced on a regular basis.

System Limitations

Electrocoagulation requires a significant amount of electricity and can increase the temperature of the effluent. Electrocoagulation is unable to treat water with high acidity or sulfates and therefore will require pre-treatment at most mine sites.[128] Metals such as copper and zinc can be effectively removed, but other metals may require more lengthy treatment times. In addition, it has been found that small quantities of metals may re-enter the treated water stream, reducing the overall efficiency.

Costs

Information on full-scale costs is not available. According to the ITRC, anticipated design costs could be high.[129] However, a potential cost savings may be realized due to less sludge generation and associated disposal costs.

Effectiveness

Electrocoagulation has not been applied for the full-scale treatment of mining wastes. The technology's inability to treat acidity or remove sulfate from sulfate-impacted waters, and the long retention times required to remove metals such as chromium and silver make full-scale implementation of the technology challenging. However, metal removal rates in pilot studies indicate that electrocoagulation may be able to achieve regulatory limits.[130] The wastewater also

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[126] ITRC, 2010.

[127] CH2M Hill, 2010.

[128] ITRC, 2010.

[129] ITRC, 2010.

[130] Rodriguez et al., 2007.

had a low pH (4.3) and contained elevated sulfates (560 mg/L). The results of the experiment were favorable, with an increase in pH to 7 and high metal removal efficiencies (copper = 99.9%, aluminum = 97.7%, manganese = 99.7%).

Properties of the MIW being treated, including conductivity, pH, chemical concentrations and particle size, affect the efficiency of the electrocoagulation treatment process. Heavy metals in water such as arsenic, cadmium, chromium, lead, nickel and zinc are generally reduced by 95 to 99 percent.[131]

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[131] Golder, 2009.

Technology: Ion Exchange

Technology Description

Ion exchange is the reversible exchange of contaminant ions with more desirable ions of a similar charge adsorbed to solid surfaces known as ion exchange resins. The active process provides hardness removal, desalination, alkalinity removal, radioactive waste removal, ammonia removal and metals removal. Similar ion exchange resins can remove selenate and selenite ions from water. Depending on the type of water that is to be treated, selective metal recovery may be an option.

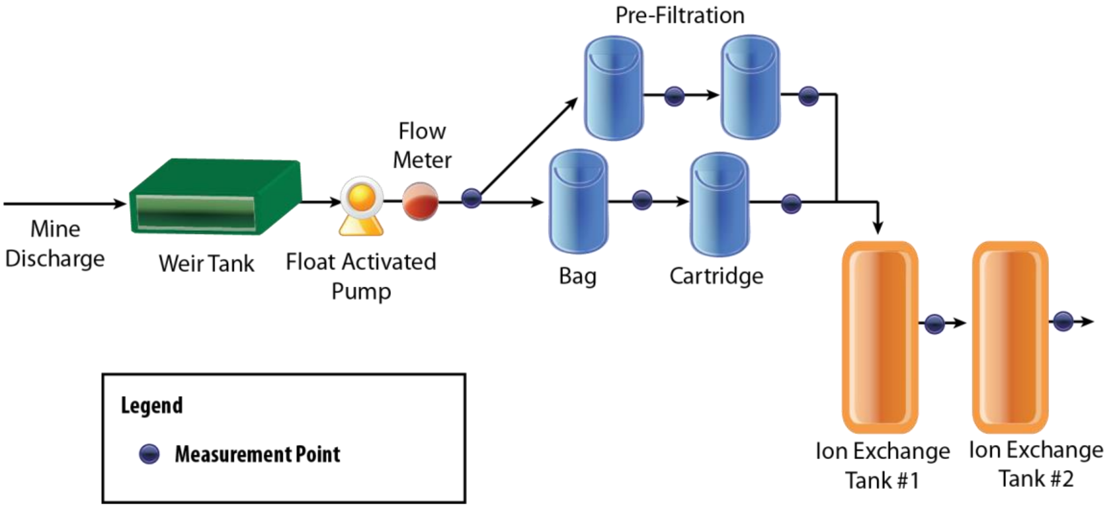
Constituents Treated

Ion exchange can remove hardness, alkalinity, radioactive constituents, ammonia and metals.

Operations

In ion exchange, ions in the water are exchanged for more desirable ions as the water passes through ion exchange resins (Figure 18).[132] There are typically four steps in a complete ion exchange process cycle: service, backwash, regeneration and rinse. The quantity of service water and backwash water generated can be significant and may require on-site storage tanks.

Figure 18: Ion Exchange Process Flow Diagram



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[132] ITRC, 2010.

When selecting the suitability and design of an ion exchange system, important considerations include the type of resin, the volume and type of regenerant, the backwash water source, backwash quantities, the need for pre-filtration of solids, column configuration, the need for pH adjustment before and after ion exchange, and cycle length.

There are various types of ion exchange resins available. It is important to work with an ion exchange resin manufacturer to conduct tests to select the appropriate resin for the contaminated source water. The higher the concentration of total dissolved solids in the water, the greater the concentration of competing anions and, therefore, the more frequently the resins will need to be regenerated.

Long-Term Maintenance

Once the ion exchange sites on the resin are completely full, the resin must be regenerated in order to be used again. A regenerant, such as sodium hydroxide solution or mineral acid, is used to remove the metals from the resin and replace them with the exchangeable ion.[133] Following regeneration, the solution must be treated and then disposed of. Treatment of the regenerant could include evaporation and crystallization, biological treatment or ZVI treatment. Resulting wastes may be classified as hazardous waste and require proper disposal.

Resins can also become clogged with suspended solids such as copper, requiring removal of ion exchange tanks before the full chemical removal capacity of the resin can be used. If this occurs, overall treatment cost can increase significantly.

Pre-treatment of the water prior to ion exchange is generally required. Additionally, exchange sites generally function at varying pHs, sometimes requiring pH adjustment to optimize resin performance. Scale removal may also be required to prevent resin fouling. The higher the concentration of total dissolved solids in the water, the more frequently the resin will need to be regenerated and rinsed with backwash water.

System Limitations

Typical limitations associated with the ion exchange technology include:

- Suspended solids will plug the resin bed and increase head loss.
- Organics, strong oxidants and high temperatures can degrade the resin.
- High sulfate is a concern and can result in exhaustion of the resin.
- Ion exchange capacity for selenium can be greatly reduced by competing ions (e.g., sulfates, nitrates).
- Resins may need to be disposed of if they cannot be regenerated, meaning high disposal costs.
- Suspended solids removal is required to reduce fouling potential.

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[133] CH2M Hill, 2010.

Costs

Annual operation and maintenance costs for a 1 million gpd system are estimated at $4.3 million (2013 USD).[134] Estimated annual costs for one site in Minnesota, the Soudan Mine discharge (average flow of 60 gpm), are about $168,993 (2013 USD).[135] The annual operation and maintenance cost estimated for selenium removal in a 700 gpm (about 1 million gpd) stripped sour water treatment system was $13.5 million (2013 USD), with over half the cost based on spent regenerant disposal.[109] The primary operation and maintenance cost was associated with regenerating or replacing the resins. At the Soudan Mine, the ion exchange tanks cost $10,888 (2013 USD) per replacement[113].

Effectiveness

Ion exchange has been successfully tested on wastewater from mining operations. The technology was pilot tested for treatment of gold heap leach solution effluents in 2005.[136] A laboratory test used process solutions from the Kennecott Mining Company on a silica polyamine resin made from polyethyleneimine impregnated with zirconium. The process solution contained 0.93 mg/L selenium and 80 mg/L sulfate at a pH of 4. The resin removed selenium to less than 1 µg/L.

In general, ion exchange works best for waters in the pH range of 4 to 8 with low suspended solids and low concentrations of iron and aluminum. The more complex the mixture, the harder it is to remove all metals effectively. The capacity of any resin to remove contaminants is limited by the type of resin, the number of available exchange sites and the input water chemistry. Capacity is generally estimated in pounds of contaminant removed per cubic foot of resin.

The Western States Petroleum Association performed a laboratory-scale investigation to remove selenium from stripped sour water effluent.[137] Selenium concentrations in the influent were up to 4,870 µg/L, and were inconsistently treated to below 50 µg/L. There is some uncertainty regarding performance for selenium removal to low levels (i.e., 5 µg/L or less), given the amount of test data. However, in the right application, ion exchange is capable of treating to 5 µg/L or less with proper resin selection and system design with consideration to water chemistry. This technology generally provides greater than 90 percent recovery rates given resin specificity for target constituent and regenerant and backwash requirements.

Ion exchange is in use at the Soudan Mine in Minnesota. The average mine dewatering discharge is around 60 gpm and contains copper and cobalt in excess of the permit standards.[138] The pH is around 4, with copper concentrations ranging from 3 to 30 mg/L, and cobalt concentrations

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[134] CH2M Hill, 2010.
[135] ITRC, 2010.
[136] Golder, 2009.
[137] CH2M Hill, 2010.
[138] Eger, 2010.

ranging from 0.1 to 0.3 mg/L. In 2003, an ion exchange unit using a selective action ion exchange resin from Siemens Corporation was installed to treat the water. Both copper and cobalt have been reduced to less than 0.01 mg/L, with concentrations of about 0.002 mg/L when the resin is new and working properly. Performance decreases as the resin loads with metal.

Figure 19: In-Mine Ion Exchange Treatment System at the Soudan Mine

Image Source: http://www.itrcweb.org/miningwaste-guidance/to_ion_exch.htm.

A full-scale system designed to treat 150 gpm was installed to treat the flow from the mine.[139] Data suggest that the resin tanks removed some of the finely suspended copper as well as dissolved copper and cobalt. Although effective, the system could not operate outside during the winter. It was moved several hundred feet to the east and an insulated, heated area was built to accommodate the ion exchange tanks. Since winter flows are much less than summer flows, surge capacity was reduced to 10,000 gallons. Metal removal is still acceptable, but the tanks are plugging due to suspended copper almost twice as fast as during the summer.

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[139] Eger, 2010.

Technology: Biological Reduction

Technology Description

Biological reduction is a process to remove elevated levels of selenium, metals and nitrate found in wastewater streams near mining operations, coal-fired power plants and other industries. This system falls broadly into the category of active BCR systems. The system is a low-energy system and uses biofilters seeded with selected strains of naturally occurring nonpathogenic microorganisms to produce treated effluent that meets or exceeds regulatory standards for selenium removal. The microorganisms in anaerobic bioreactors reduce selenium (in the form of selenite and selenate) to elemental selenium. The end product is a fine precipitate of elemental selenium that is removed from the bioreactor during periodic backflushing. Several pilot-scale tests have proven the reliability of this process. It is now operational full scale at several locations worldwide.

Constituents Treated

Biological reduction removes elevated levels of selenium, nitrate, mercury and metals in wastewater.

Operations

GE's Advanced Biological Metals Removal System® (ABMet) technology, a type of fixed bed BCR, uses special strains of common nonpathogenic microbes that facilitate the conversion of soluble selenium into elemental selenium, which is removed from the system during periodic backwashing (Figure 20). The microbes, which are fed the molasses-based nutrient, are seeded in a bed of activated carbon that acts as a growth medium for the microbes to create a biofilm. Selenium-laden wastewater passes through a low-power, fixed-bed bioreactor and a reduction reaction occurs. Other than the addition of the nutrient, the system is designed to be self-sustaining once established. Pre-treatment will likely be a site-specific consideration for this technology.

Figure 20: ABMet® Flow Diagram

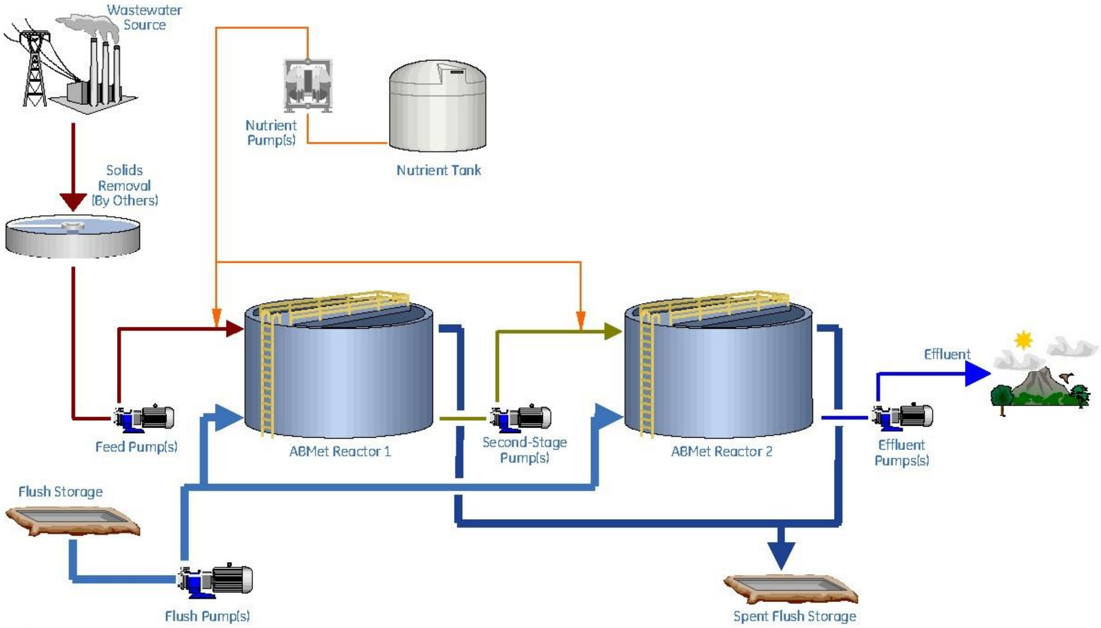

Image Source: GE 2009.

ABMet® systems are intended to be scalable and allow for the accommodation of any flow and future system expansions. The systems can be custom designed for specific needs or pre-engineered as modular units for lower cost and quicker turnaround times. Installation with modular systems containing prefabricated bioreactor tanks and skid-mounted pump-and-control packages allows for a short execution timeline from ordering to a fully operating system.
The systems can be housed in a structure or insulated for outside installation and operate over a broad temperature range (4^{o} to 40^{o} C). Preheating of the water is not required.

Example Sites

The ABMet® system has been demonstrated at both pilot and full-scale operational levels in mining influenced and flue gas desulfurization wastewaters. ABMet® was an EPA pilot demonstration technology at Kennecott Utah Copper Corporation's Garfield Wetlands-Kessler Springs site in Utah.[140] Garfield Wetlands-Kessler Springs water was pumped to the system at a flow rate of about 1 gpm using a solar pump. A flow meter/totalizer recorded the actual flow rate and the total volume of water processed by the system. The water then entered a series of 500-gallon bioreactors containing a carbon/biosolids/biofilm combination or carbon/biofilm, depending on the test series. Nutrients were supplied to the reactors at three locations in the process. When the water had flowed through the appropriate number of bioreactors, it was filtered by a slow sand filter prior to discharge. During six months of operation, from October

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[140] MSE, 2001.

1999 through April 2000, the ABMet® system was able to lower the concentration of selenium from 1,950 µg/L to below 2 µg/L, with hydraulic retention times as low as 5.5 hours.

The ABMet® system was also pilot tested at a coal mine in West Virginia using flow rates of 1, 2 and 3 gpm.[141] The system achieved selenium removal from influent of 43 µg/L to less than 5 µg/L. A full-scale ABMet® system was installed in 2002 at a gold mine in the Black Hills of South Dakota. The system was designed to accommodate average flows of 100 gpm and a maximum flow of 300 gpm and to remove selenium from 100 µg/L to below 5 µg/L.[142]

Long-Term Maintenance

GE's ABMet® system is designed to be relatively self-sustaining, requiring only the addition of a proprietary molasses-based nutrient.[143] It does not require ongoing replenishment of microbial cultures or granular activated carbon. Primary treatment consists of pH control so that neutral pH is achieved for biological treatment and total suspended solids removal to prevent clogging of the activated carbon media.

ABMet® systems generate wastes that require proper waste handling and management. The low solids volumes generated during the process are mostly elemental metals, metal sulfides and some biomass.[144] They can be dewatered with a belt or plate/frame press, typically to a 15 to 30 percent dry product. Waste products can also be sent to thickening/dewatering, settled or recirculated to a clarifier. Biological residuals must be thickened and dewatered for landfill disposal and backwashing may be required to periodically slough off excess microbial growth and prevent short-circuiting of flow, as well as for degassing. Carbon dioxide and nitrogen gases accumulate in the bioreactor cells over time from biological reduction. The trapped gases decrease the flow path and increase head losses through the system. Therefore, the bioreactor cells are backwashed for degassing. Additional backwashes are performed on a less frequent basis to remove precipitated solids and excess biomass.

System Limitations

The footprint of the ABMet® system is dependent on the flow to be treated. The overall treatment footprint can be larger if pre-treatment or post-treatment is required. Disadvantages of the system include the need for pre- and post-treatment steps to remove suspended solids, backwashing to prevent plugging and short-circuiting of flow, and temperature dependence. In addition, capital costs can be high due to the large volumes of granular activated carbon required.

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[141] CH2M Hill, 2010.
[142] CH2M Hill, 2010.
[143] GE, 2013.
[144] GE, 2009.

Costs

In 2010, GE estimated that the estimated total installed cost for a 1 MGD system would be about $32.1 million (2013 USD), with an annual operation and maintenance cost of approximately $3.2 million (2013 USD).[145] The system requires low operation and maintenance costs of $0.11 to $0.54 per 1,000 gallons of water treated. Systems can be custom designed for specific needs or pre-engineered as modular units for lower cost and quicker turnaround times.

The Garfield Wetlands-Kessler Springs site's cleanup system required a capital cost of $846,889 (2013 USD), with an annual operating and maintenance cost of $189,329 (2013 USD). In 2001, the system costs were $1.74 (2013 USD) per 1,000 gallons treated. The capital costs included biofilm support materials, inoculum, system design, building modifications, equipment purchase and installation, construction, commissioning, and project closeout. The annual operation and maintenance costs included nutrient costs, manpower, maintenance and power for equipment use.

Effectiveness

Current ABMet® systems are treating flow rates as low as 5 gpm and as high as 1,400 gpm. The systems are discharging a treated effluent containing 5 µg/L of selenium or less based on incoming water characteristics and process controls. The system has proven its capability to attain complete nitrate-nitrogen removal, and is able to handle selenium loads from 10 µg/L to 10,000 µg/L. Custom designs are also available to meet specific site conditions or large flows of more than 1 MGD.

Table 12: ABMet® System Capabilities

Constituent	Units	Typical feed	Effluent
Nitrate-N	mg/L	10-250	ND<1
Selenium	µg/L	<10,000	<0.005
Mercury	µg/L	<5	<0.012[1]
Other Metals	µg/L	<10	<0.01

Source: http://www.gewater.com/pdf/events/2009/power_ind/GE-ABMet.pdf.

1. With additional treatment processes, depending on specific water chemistry.

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[145] GE, 2010.

Technology: Ceramic Microfiltration

Technology Description

Ceramic microfiltration is a pressure-driven membrane separation process designed to remove heavy metals from acid mine drainage. It uses ceramic membrane microfilters to remove precipitated solids that allow for effective cleaning to restore membrane permeability. Ceramic microfiltration requires significant energy input and therefore has high operating costs. However, it may be more cost effective than other alternatives since it may replace several unit treatment processes with a single process.

Figure 21: Ceramic Microfiltration System at the Upper Blackfoot Mining Complex, Montana

Image Source: http://deq.mt.gov/StateSuperfund/UBMC/watertreatmentplant.mcpx

Constituents Treated

Ceramic microfiltration captures precipitated heavy metals, including chromium, nickel, lead, uranium, iron, manganese, selenium, copper, cobalt, cadmium and zinc.

Operations

A ceramic microfiltration water treatment plant was built in 2009 at the Upper Blackhawk Mining Complex near Lincoln, Montana, to treat source water flows from adit discharges and seeps.[146] The plant incorporates ceramic microfiltration technology primarily to remove cadmium, copper, manganese and zinc. The design flow to the plant is 168 gpm, with an expected average flow of 63 gpm.[147]

Prior to entering the ceramic microfiltration system, the source waters are collected and conveyed to the feed tank to oxidize iron and reduce the levels of carbon dioxide. The feed tank water is pumped through two neutralization tanks and a concentration tank to remove suspended solids.[148] This pre-treated water is pumped through the ceramic microfiltration modules, where metal-bearing suspended solids are filtered out of solution. Filtered water is sent to the pH adjust tank and solids are returned to the concentration tank and pumped to the sludge tank. In the pH adjust tank, sulfuric acid is added to the treated water to adjust the pH back to between 6 and 9. The treated water is then discharged to the Upper Blackfoot River through a discharge pipe. Solids in the sludge tank are allowed to settle and are periodically pumped to a sludge press for dewatering. The dewatered sludge is a non-hazardous waste and is stored temporarily at the treatment plant.

Figure 22: Ceramic Microfiltration Cross-Flow Diagram

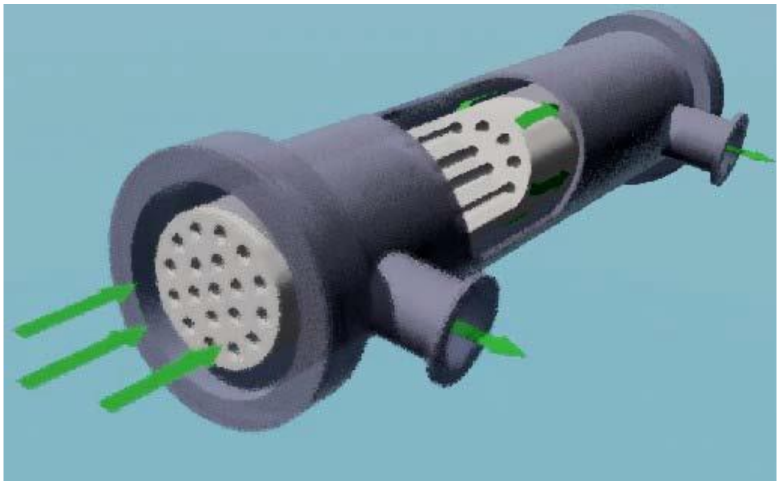

Image Source: http://www.epa.gov/nrmrl/std/mwt/annual/annual2002/mwtp2002.pdf.

Long-Term Maintenance

Ceramic microfiltration systems can include a backpulse system that periodically forces a small amount of filtered water in reverse through the membrane to remove built-up solids from the membrane surface. This backpulse function prolongs the cycle life between chemical cleanings. Chemical cleaning is required at a one-week-to-three-month interval depending on the system and the MIW.[149] Backwash waste requires disposal or recycling; chemical waste is generated during periodic cleanings.

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[146] MT DEQ, 2013.

[147] MT DEQ, 2013.

[148] MT DEQ, 2013.

[149] CSM, 2009.

System Limitations

The biggest technical challenge with the use of membrane systems is the fouling due to buildup of colloids, soluble organic compounds and microorganisms that conventional pre-treatment methods typically do not remove well. Frequent membrane cleaning may be required, leading to reduced efficiency and a shorter membrane life. In addition, residuals may be classified as hazardous waste, requiring higher-cost disposal.

Costs

The operating cost of the system at the Upper Blackfoot Mining Complex was $666,192 (2013 USD).[150] The system's capital and operation and maintenance costs were significantly lower than the estimated cost of an alternative clarifier system.

The U.S. Department of Energy and EPA's Mine Waste Technology Program conducted a pilot-scale test of a ceramic microfiltration system at the Central City/Clear Creek Superfund site in Idaho Springs, Colorado, in December 2002.[151] The evaluation indicated that the system successfully removed heavy metals while operating at 300 gpm to 500 gpm. However, a full-scale ceramic microfiltration system was not installed.

Effectiveness

The water treatment plant at the Upper Blackhawk Mining Complex currently meets discharge limits for all required metals, removing about 99.5 percent of all metals in the treated water.[152] The system operates at flow rates of 25 to 135 gpm. Sludge is classified as non-hazardous, which allows for disposal of solids on site. From the initial concept to completion, the project took nine-and-a-half months. Construction modifications to meet final discharge began at the water treatment plant in the spring of 2011 so that the plant could effectively treat the millions of gallons of metals-laden water seeping from the historic mines. The plant's sludge generation process has been modified so that material extracted from the water does not require additional treatment prior to landfilling off site.

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[150] Stewart Environmental.

[151] EPA, 2002b.

[152] MT DEQ, 2013.

Appendix A. Summary of Treatment Technologies

This table includes summary information for the technologies discussed in the body of the report, as well as additional technologies or products designed as passive or low-cost treatment options. The mention of trade names, specific vendors or products does not represent an actual or presumed endorsement, preference or acceptance by EPA or the federal government. Stated results, conclusions, usages or practices do not necessarily represent the views or policies of EPA.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Adsorption To Peanut Shells	Innovative technology using carbonized peanut shells to physically adsorb selenium.	Selenium	Lab-scale	None	Before treatment, peanut shells are treated with strong sulfuric acid to carbonize the shells while partially oxidizing the cellulose and hemicelluloses and fragmenting the lignin. The sulfuric acid treatment results in a carbonaceous material with functional groups for both the sorption and reduction of selenium.	Unknown	Sorption was found to be temperature dependent.	This process has only recently been developed and is not well characterized. Peanut shells are readily available at low cost.	Removals as high as 63 percent were observed for 25 mg/L selenide solutions. Selenite sorbed to the material at an optimal pH of 1.5. As pH increased, sorption capacity decreased.
Algal-Bacterial Selenium Removal	Algal treatment occurs via enhanced cyanobacterial and algal growth through nutrient addition. These additional nutrients increase algal biomass, thereby increasing selenium volatilization rates. In addition to treatment by volatilization, the algal biomass generated also serves as a carbon source to support microbial selenium reduction processes.	Selenium	Pilot-scale	Panoche Drainage District on the west side of the San Joaquin Valley, California	This system uses a combination of ponds containing algae and bacteria in which selenate is reduced to selenite and elemental selenium.	Unknown	Seasonally limited with treatment affected by duration of solar light and ambient temperatures. Difficulty of generating sufficient biomass to promote biological reduction and the inability to treat selenium to regulatory levels. Although much of the total selenium may be removed using the ABSR system, it is essential that the remaining selenium is in a form that is not readily bioavailable to aquatic organisms. In the ABSR system, if the selenium that remains has been transformed into a more bioavailable form, then the system could possibly be increasing the concentrations of selenium absorbed by aquatic life.	Potentially low-cost treatment, but can require large land area as well as the need for separation of the high-rate and reduction ponds. The preliminary total cost estimate for a 10-acre-foot per day ABSR facility is less than $291 (2013 USD) per acre-foot of treated drainage water. The ABSR system is one of the most economical and therefore easily adopted selenium removal systems.	Preliminary results have shown that the method may be able to reduce the total selenium in the drainage water up to 80 percent. During 1997 and 1998, the best-performing ABSR plant configuration reduced nitrate by more than 95 percent and reduced total soluble selenium mass by 80 percent. The results of this study suggest that the ABSR system may not be successfully reducing the bioavailability of selenium to aquatic organisms. Although microcosm data was limited, results lead to the conclusion that certain steps of the system may be increasing bioavailability.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Alkalinity-Producing Systems	Alkalinity-Producing Systems (APS) combine the use of an anoxic limestone drain (ALD) and anaerobic compost wetlands. Ponded water overlies an 18-inch layer of organic material, usually compost, which is over an 18- to 24-inch layer of limestone. Acid water is ponded over the materials and the head created by the column of water forces the water through the organic material to filter out or precipitate ferric iron and to consume oxygen through organic matter decomposition.	Acidity, copper, lead, zinc, ferrous iron, manganese	Full-scale	The Douglas Highwall site Tucker County, West Virginia The Copper Basin Mining site, southeastern Tennessee	Under current designs, ponded water about 3 to 6 feet in depth overlies an 18-inch layer of compost, which is over an 18- to 24-inch layer of limestone. Douglas Highwall site: The treatment system was constructed by digging a large trench along the old railroad grade. The first cell of the system was 1,225 feet long by 8 feet wide and 6 feet deep. About 2 feet of gravel-sized limestone were placed on the bottom with 4 feet of organic material (a peat, hay and soil mixture, 50:40:10) over the top. Cell I was filled with about 880 tons of limestone and about 1,450 cubic yards of organic material.	Routine maintenance is typically limited to inspection of the surface for evidence of leakage in the anoxic cover material, and periodic cleaning of the discharge point to remove accumulated iron oxides. The systems are generally designed for limestone replenishment every 15 to 25 years, depending on the characteristic of the drainage flow. Flushing the wetlands may be a solution to increasing the treatment success and may aid in the prevention of clogging.	In situations where dissolved oxygen concentrations are >1 mg/L, the oxygen must be removed from the water before introduction into an anoxic limestone bed. For waters with high sulfate (>1,500 mg/L), gypsum may also precipitate. Noxious odors (hydrogen sulfide) are sometimes produced near the system.	The entire project cost at the Douglas Highwall site, including all reclamation activities and water treatment systems, was $2.2 million (2013 USD). The wetland-drain system costs were about $630,489 (2013 USD). The cost to treat the acid mine drainage with this wetland system is estimated to be $1,305 (2013 USD) per ton of acid neutralized. This cost per ton is almost equivalent to sodium hydroxide chemical treatment cost, which is estimated at $1,349 (2013 USD) per ton.	The alkaline-producing systems are predicted to remove about 15 to 20 gallons per square meter per day of acid. This value represents the commonly accepted target that most wetland builders expect.
Aluminator® Passive Treatment System	The Aluminator© is an adaptation of a limestone drain in which aluminum hydroxide will accumulate for recovery.	Aluminum, iron, acidity	Pilot-scale	Metro site, Pennsylvania Casselman River Watershed, Somerset County, Pennsylvania The Buckeye Reclamation Landfill site, Belmont County, Ohio The Little Mill Creek site, Jefferson County, Pennsylvania The Greendale site, Clay County, West Virginia	Mine drainage enters the Aluminator© on the surface of the treatment unit and flows downward through the treatment column. A standing pool of water provides a buffer from flow surges, allows for a relatively even distribution of flow across the entire treatment area, and provides a positive head, essentially forcing the water down, into, and through the underlying substrates.	These systems require more operation and maintenance (O&M) than either aerobic wetlands or ALDs. Treatment effectiveness can be maintained by periodically flushing the system.	Efficiency can decrease with sustained or uncontrolled high flow events. There has been some decline in effectiveness of some of the systems over time but mainly in sites treating water with significant levels of aluminum and a low pH.	The total cost of the Metro site's M1 system was $217,636. The cost of the M2 system was $131,901 (2013 USD).	Metro site systems treated: Aluminum from 90/110 mg/L to 20/25 mg/L. Iron from 270/290 mg/L to 140/170 mg/L. pH from 2.8/2.7 to 5.8/5.8. Alkalinity from 0 to 90 and 100 mg/L CaCO3.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Anoxic Limestone Drains (ALD)	ALDs are a simple treatment method – buried limestone in air-tight trenches intercepts acidic discharge water. ALDs are used to generate alkalinity and must be followed by ponds and aerobic wetlands that oxidize and remove the dissolved metals.	Acidity	Full-scale	Fabius Coal Preparation Plant, Alabama Copper Basin Mining site, Tennessee Hartshorne/Whitlock-Jones site, Hartshorne, Oklahoma Ohio Abandoned Bituminous Coal, southeast Ohio Tecumseh - AML site 262, Indiana Tennessee Valley Authority site, Alabama Valzinco Mine, Virginia	Construction of an ALD consists of a trench containing limestone encapsulated in a plastic liner and covered with clay or compacted soil to maintain anoxic conditions, as well as to prevent water infiltration and to keep carbon dioxide from escaping. The width and length of the trench are based on the levels of dissolved metals present in the mine drainage, the retention time needed to raise the pH, as well as the amount of area that is available for construction. The ALD may be capped with topsoil and vegetation to control erosion. The two factors that must be considered when sizing an anoxic limestone drain are the accommodation of the maximum probable flow and the desired longevity of the drain.	Routine maintenance is typically limited to inspection of the surface for evidence of leakage in the anoxic cover material, and periodic cleaning of the discharge point to remove accumulated iron oxides. The systems are generally designed for limestone replenishment every 15 to 25 years, depending on the characteristic of the drainage flow. Maintenance costs for ALDs are not expected to be significant. Apart from monitoring costs, costs should be limited to periodic inspection of the site and maintenance of the vegetation cover.	Metal removal must occur elsewhere to prevent clogging of the bed and system failure. ALDs must be kept anoxic to prevent the oxidization of soluble ferrous iron to the insoluble ferric species. Although ALDs are documented to have success in raising pH, the differing chemical characteristics of the influent mine water can cause variations in alkalinity generation and retention of metals. Most ALD systems exhibit reduced effectiveness over time and eventually require maintenance or replacement. Use of ALDs as a standalone treatments system might not achieve effluent compliance limits.	The cost of installing ALDs can vary, depending largely on location and chemical makeup of the influent. Operators of the Tennessee Valley Authority abandoned mine site in Alabama reported that their capital cost was about $0.27 per 1,000 gallons of water and their O&M costs were about $0.11 per 1,000 gallons of water (2013 USD). A typical ALD at most locations in Canada is expected to cost in the range of $6,000 to $37,000 (2013 USD), depending on chosen dimensions and design flow. This estimate would not apply to more remote sites or sites where an ALD would require extensive excavation or blasting.	Where influent mine water contained less than 1 mg/L of both ferric iron and aluminum, ALDs produced consistent concentrations of alkalinity for over 10 years. Alkalinity concentrations in the effluent range from 80 mg/L to 320 mg/L as CaCO3, with near maximum levels reached after about 15 hours of detention in the ALD. An ALD receiving influent mine water containing 21 mg/L of aluminum failed within 8 months due to clogging.
Aquafix	The Aquafix system uses the recognized effectiveness of lime addition to raise the pH of Mining-Influenced Water (MIW) to precipitate metals.	Aluminum, copper, iron, manganese, zinc	Pilot-scale	Summitville Mine, Colorado Almeda Mine, near Grants Pass, Oregon Dinero Tunnel, Colorado Crystal underground copper mine, Butte, Montana Jennings Randolph Lake, Maryland	Water passing through the water wheel drives the auger, which distributes lime pellets into the stream at a fully adjustable rate, to ensure precise treatment levels. In a pilot study, a rock drain downstream from the treatment unit promoted dissolution of the lime. Effluent from the rock drain was further aerated in a mixing tank and subsequently sent to two settling tanks connected in series. The calculated total residence time of the system was about two days, but the flow rate in this system tended to fluctuate, affecting residence time.	Requires regular inspections to ensure that proper flow is maintained through the treatment systems. Aquafix units do not require constant monitoring. Units can operate continuously for many days without human attention.	At the Colorado site (Dinero Tunnel), the inlet hose became clogged with iron hydroxides (yellow boy), which reduced the flow and the lime dispensing rate. Revolutions per minute and pH values were consistently lower than expected. It became necessary to disconnect the hose each week and flush out the sludge. Operational problems also were encountered in Oregon with accumulation of the granular lime below the dispenser.	The average cost for a permanent Aquafix system designed to treat 25 gallons per minute (gpm) is expected to be about $89,531 (2013 USD) per year, based on a 15-year system life. Jennings Randolph Lake: Cost for treating this water is about $50,000 per year, or a little more than $0.02 per 1,000 gallons of water. A portable Aquafix unit costs about $21,582 (2013 USD).	At the Oregon site, metal concentration reductions ranged from 94 percent to 99 percent for the principal acid mine drainage metals of aluminum, cadmium, copper, iron, lead, manganese and zinc. Removal efficiencies: Al - 97%, Cu – 99%, Fe - 99%, Mn - 97% Zn - 99%

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Bauxsol^{TM}	A material based on mud residues generated during alumina production has the capacity to neutralize acid and trap trace metals, with application to the treatment of acid rock drainage and mine tailings. Bauxsol^{TM} is prepared by chemical and physical modification of the caustic red mud residue generated by the Bayer process for extracting alumina from bauxite prior to electrolytic reduction. Pure Bauxsol^{TM} has a high acid-neutralizing capacity due to the abundance of amorphous and finely crystalline mineral phases that form weak bases. Pure Bauxsol^{TM} also has a very high trace metal trapping capacity. It also has a high capacity to trap and bind phosphate and some other chemical species. Reagents precipitate and settle within 48 hours to form a thin layer of sediment.	Acidity, cadmium, copper, zinc, aluminum, iron, lead, nickel, and zinc, arsenic, cyanide, phosphates	Full-scale	Gilt Edge Mine, South Dakota	The treatment involves directly adding product(s) to the dammed water. The products can be dispersed into the dam using any conventional means and usually by existing infrastructure on site. The duration of the treatment, the selection of the product or products to be used, the intervals between applications of the various products will be determined by the degree of contamination and the size of the dam. After treatment, suspended particles in the water in the dam quickly settle to form a thin layer of sediment typically less than 5 millimeters thick. The extracted metals remain locked in the exhausted Bauxsol^{TM} sediment covering the tailings.	The technology allows for treated water to be safely discharged into the environment, obviating the need for indefinite storage of contaminated water. The sediment remaining after the treated water is discharged is able to be easily revegetated and will support healthy plant growth. Therefore, it is expected that there is no added requirement to treat or dispose of this sediment.	For effective treatment, the arsenic should be present as arsenate. If arsenite is present, it should be oxidized before treatment with Bauxsol™. For ponded water, a good solution is to use additives, such as ferrous sulfate, ferric chloride, aluminum sulfate or jarosite minerals to create more positive charges on the surfaces of mineral particles in the Bauxsol™. Bauxsol™ does remove arsenite, but the efficiency is only about 10 percent of that for the removal of arsenate. Arsenic uptake by Bauxsol™ is interfered with by anions such as phosphate and sulfate, but the interference is not serious.	Bauxsol^{TM} is relatively inexpensive and can occur in situ. There is no requirement to invest significant amounts of capital in treatment and process plants. Bauxsol^{TM} technology can be cost effective because it uses one waste technology to treat another and is not capital-intensive. The overall cost associated with using the process for any site is affected by the quantity of each product required for the treatment, which depends on the level of contaminants and the quantity of water, and the location of the site relative to the production facility. It is expected that most water bodies can be treated in the price range of $1,298 to $2,597 (2013 USD) per 1 million liters. However, specific costing is required on a job-by-job basis.	Bauxsol^{TM} is designed to sequester over 99.99 percent of all heavy metals from soils and water, including acid, arsenic, cyanide and toxic metal combinations. Its acid neutralization capacity is also high, due to the abundance of amorphous and finely crystalline mineral phases that form weak bases. Neutralizes 3.5 to 7.5 moles of acid/kg Bauxsol^{TM} (14 moles/kg if the pH is < 5). The ability of the minerals to trap trace metals is also strongly time-dependent. Although most of the initial metal trapping is complete within 24 hours, metal trapping will continue, albeit more slowly, for many months and the longer the material is left, the more tightly the metals are bound.

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Biochemical Reactors (Bioreactors)	Biochemical reactors (BCRs), or bioreactors, treat MIW by using microorganisms to transform contaminants and to increase pH in the treated water. BCRs can be designed as open ponds and buried ponds or within tanks or even in trenches between mine waste and a surface water body. The most commonly used BCRs for treating mining influenced water are operated anaerobically. They are also called "sulfate-reducing" bioreactors.	Selenium, cadmium, copper, nickel, lead, zinc, arsenic, chromium	Full-scale	West Fork Mine, Missouri Leviathan Mine, California Golinsky Mine, Shasta Lake City, California Stowell Mine, Shasta Lake City, California Copper Basin, Tennessee, Central City/Clear Creek Superfund Site	The design of BCRs is controlled by the site-specific MIW characteristics of pH, flow, temperature and the type and concentration of metals. Application requires bench-scale and pilot-scale testing to estimate site- and effluent-specific parameters. Sulfate-reducing bioreactors typically require large amounts of organic materials that are usually considered waste. Enhanced sulfate-reducing bioreactor cells can consume liquid organic wastes like antifreeze or cheese whey.	Minimal maintenance is required for bioreactors, but performance can decline due to clogging. Since the passive systems typically do not require any external power and can operate without continual maintenance, they are attractive for remote and abandoned sites.	BCRs require a large footprint. Substrate degrades over time. BCR design is influenced greatly by available space, since the water to be treated must reside within the BCR for a certain period of time, called retention or residence time. The most important mixture component is the organic carbon source. Several studies conducted to find the best mixture of natural organic substrates for sulfate-reducing bioreactor showed that a combination of organic sources is preferable over a unique source.	Bioreactors should have lower O&M costs than standard chemical treatment approaches. Bioreactors can be costly to construct since generally the systems are lined and often contain additional components, such as settling ponds and/or aerobic polishing cells. As constructed, the Leviathan Mine system requires 0.75 acres. The capitol costs for construction of the gravity-flow operation amounted to $1,062,100 (2013 USD) and changing to the recirculation mode added nearly $38,000, for a total of approximately $1.1 million (2013 USD). Operating at an average flow rate of 10 gpm, the O&M costs of the system are $19.51 (2013 USD) per 1,000 gallons of treated water.	Various types of media for gravel pit seepage can result in 98 percent removal of selenium and can achieve less than 5 micrograms per liter of selenium. Pilot and field-scale passive bioreactors filled with mixtures of organic and cellulosic wastes were installed in Canada and the United States and efficiently removed sulfate and metals for periods up to 5 years. Manganese and arsenic are less efficiently removed as sulfides in passive bioreactors.
Biological Reduction Of Selenium (BSeR™ And GE's ABMet®)	This process uses specially developed biofilms that contain specific proprietary microorganisms in anaerobic bioreactors to reduce selenium (in the form of selenite and selenate) to elemental selenium. The end product is a fine precipitate of elemental selenium that is removed from the bioreactor with backflushing.	Selenium, mercury, complete nitrate removal and removal of a variety of other metals	Full-scale	Kennecott Utah Copper Corporation Garfield Wetlands-Kessler Springs Kennecott North site, Utah	ABMet® (formerly BSeR™) is a plug-flow, anaerobic bioreactor in which a selenium-reducing bacterial biofilm is supported on granular activated carbon. The system is inoculated with a mixture of proprietary and indigenous microorganisms, and reducing conditions are maintained by feeding a molasses-based nutrient mixture to the system. Systems can be custom designed for specific needs or pre-engineered as modular units for lower cost and quicker turnaround times.	The ABMet® system is designed to be relatively self-sustaining, requiring only the addition of a proprietary molasses-based nutrient. It does not require ongoing replenishment of microbial cultures or granular activated carbon.	Requires pre- and post-treatment steps to remove suspended solids, backwashing to prevent plugging and short-circuiting of flow, and temperature dependence.	GE estimates that for a 1 million gpd system, the estimated total installed cost would be about $32.1 million, with an annual O&M cost of about $3.2 million (2013 USD). O&M costs of $0.11 to $0.54 per 1,000 gallons of water treated (2013 USD).	Process can treat flow rates as low as 5 gpm and as high as 1,400 gpm, while achieving up to 99 percent removal rates and discharging a treated effluent containing 5 µg/L of selenium. Nitrate (typical feed of 10-25 mg/L), possible effluent: ND<1mg/L. Selenium (typical feed of 10,000 µg/L), possible effluent: <0.005 µg/L. Mercury (typical feed of 5 µg/L), possible effluent: <0.012 µg/L.

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Catalyzed Cementation Of Selenium	Catalyzed cementation removes heavy metals from solution by cementation on an iron surface. The process is optimized by adding catalysts that increase selenium removal efficiency.	Selenium	Pilot-scale	Kennecott Utah Copper Corporation Garfield Wetlands-Kessler Springs Kennecott North site, Utah	Feed water is fed through a series of static mixers where pH is lowered before entering the elemental iron reactor. The reactor is a specialized tank designed to fluidize iron particles. The iron particles carried out are trapped in a small, cone-bottom tank and pumped back to the reactor for reuse. The processed feed water exiting the small, cone-bottom tank is routed to an 80-gallon reactor where the pH is raised again with a lime slurry and an oxidizer is added that completes the required reaction.	Solids that accumulated in the bottom of the thickener were periodically removed by a diaphragm pump. This sludge slurry was then processed by a filter press. The sludge liquid separated from the solids was returned to the thickener. The filter cake solids removed from the filter press were prepared for analysis or disposal by placing them in appropriate containers. Both filter cake samples were analyzed and found to be below the TCLP threshold value for selenium of 1 mg/L.	High chemical costs and solid waste disposal required. No long-term studies of the stability of the cementation waste have been undertaken.	Capital: $1,428,860 (2013 USD for all monetary values) Annual O&M Cost: $1,537,114 Net Present Value of Annual O&M Costs: $12,529,666 Total Net Present Value: $2,248,485 Net Present Value of $/1000 gallons treated: $10.78 Generic cost estimate: Based on 300 gpm plant, 2 mg/L selenium influent, Capital: $1.6 million, O&M: $1.6 million, Net Present Value: $12.5 million, $/1000 gal.: $10.78, $/kg selenium: $1,423.	Garfield Wetlands-Kessler Springs water: water with total selenium concentrations of 1,950 µg/L (primarily as selenate) was tested at a flow rate of 1 gpm. Even after extensive optimization in the field, the lowest effluent concentration achieved was 26 µg/L. Continued optimization in the laboratory achieved a mean effluent selenium concentration of 3 µg/L.
Ceramic Microfiltration	This treatment system is designed for the removal of heavy metals from an acid mine drainage system. It uses ceramic microfiltration to remove the precipitated solids.	Metals	Pilot-scale	Black Hawk and Central City, Colorado Kennecott Utah Copper, LLC, (Kennecott) Bingham Canyon Mine, Utah Upper Blackfoot Mining Complex, near Lincoln, Montana	Wastewater proceeds through a hydroxide precipitation step, which consists of adding sodium hydroxide. The pH will be adjusted to between 8.5 and 9.5 in a two-stage pH adjust system. The wastewater is then transferred to the concentration tank. At this point, the wastewater will be pumped through the cross-flow ceramic membrane. The absolute pore size of the membrane is 0.2 microns. Therefore, the only metals that will remain in the filtered water will be dissolved metals.	Ceramic membranes should be backwashed periodically and chemical cleaning is required at weekly to three-month intervals, depending on water quality. Backwash waste requires disposal or recycling. Chemical waste is generated during periodic cleanings.	High capital and O&M costs. Requirement of osmotic pressure. Fouling of membranes and scale production. Reliance on external power. Potential difficulty of concentrate disposal. Feed solution regarding quality predictability.	Pressure-driven membrane separation processes may involve higher capital and O&M costs than other water treatment technologies, depending on the size of the treatment unit, the volume of feed solution to be addressed, and the cleanup goals. The system at the Upper Blackfoot Mining Complex cost $666,192 (2013 USD).	Ceramic microfiltration removes 99.5 percent of the heavy metals from wastewater streams with a system that meets the new proposed standards. Upper Blackfoot Mining Complex: the system has been operating and meeting standards since January 2009.

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Colloid Polishing Filter Method (CPFM)	CPFM technology uses a proprietary compound (Filter Flow [FF] 1000) that consists of inorganic, oxide based granules. FF 1000 is formulated to remove heavy metals and radionuclides from water through a combination of sorption, chemical complexing and filtration. The technology developer, Filter Flow Technology, Inc., states that sorption on the FF 1000 accounts for the majority of the removal action.	A wide range of ionic, colloidal and complex non-tritium radionuclides and heavy metals	Pilot-scale	CPFM technology was initially demonstrated under the SITE program at DOE's Rocky Flats Environmental Technology site near Golden, Colorado. CPFM has been demonstrated independent of the SITE program at two locations at the Department of Energy Hanford facility, where it removed strontium-90, cesium-137, plutonium-239 and americium-241 from water at K-Basin, and strontium-90 from ground water at the Site 100N Area (N-Spring).	Pre-treated water is pumped from the bag filters to the colloid filter press units where heavy metals and radionuclides are removed and discharged.	The only major system components that require regular maintenance are the filter packs in the colloid filter press unit. They require periodic replacement or regeneration.	A CPFM system is not designed to operate at temperatures near or below freezing. If such temperatures are anticipated, the CPFM system and associated storage tanks should be kept in a heated shelter, such as a building or shed. In addition, piping to the system must be protected from freezing. A CPFM system requires potable water, electricity and compressed air for operation.	Ground water remediation costs for an l00-gpm CPFM system could range from about $3.23 to $11.32 (2013 USD) per 1,000 gallons, depending on contaminated ground water characteristics and duration of the remedial action. The cost of building a system is estimated to be about $121,244 to $161,658 (2013 USD). A skid-mounted system that treats water at flow rates up to 100 gpm could be built for about $242,487 to $323,316 (2013 USD).	Filter Flow Technology, Inc. reports that its CPFM system has effectively removed trace ionic heavy metals and non-tritium radionuclides from water that has been pre-treated to reduce suspended solids.

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Constructed Wetlands	Constructed wetlands use soil- and water-borne microbes associated with wetland plants to remove dissolved metals from acid mine drainage. Constructed wetlands can be designed as aerobic wetlands, anaerobic horizontal-flow wetlands and vertical-flow ponds (vertical-flow wetlands).	Iron, manganese, sulfate, arsenic, dissolved aluminum, copper, zinc, cadmium, nickel, lead, selenium	Full-scale	Burleigh Tunnel wetland, Colorado Asarco's West Fork site, Missouri Somerset wetland, Somerset County, Pennsylvania Latrobe wetland, Westmoreland County, Pennsylvania Friendship Hill wetland, Fayette County, Pennsylvania Commerce/Mayer site, Oklahoma Copper Basin site, Tennessee Keystone site, California Hartshorne/Whitlock site, Oklahoma	Design of a constructed wetland for the treatment of acid mine drainage varies based on a site's characteristics. The most important design considerations are biochemical processes, loading rate and retention time, slope, substrate, vegetation, sediment control, morphometry, seasonality, and regulatory issues.	Constructed wetlands can operate for long periods of time with minimal O&M. The concentration of contaminants must be monitored to maintain ecological health of the system. Disposal of accumulated material is required.	Requires appropriate land for wetlands construction. High initial construction cost. Sensitivity to high throughput excursions. Disposal of accumulated material. Relatively slow performance in comparison to other treatment technologies. Dependency on local climatic conditions, which may lead to reduced efficiency during colder seasons. Potential to become a permanent feature of the ecosystem, requiring long-term maintenance.	The U.S. Naval Facilities Engineering Service Center's Remediation Technology Online Help Program lists the costs of constructed wetlands treatment at between $0.15 and $1.00 (2013 USD) per gallon of water treated.	Typical range of removal efficiencies observed in wetlands constructed to treat drainage: pH: >6 for coal mine drainage and metal mine drainage. Acidity: 75-90 percent for coal mine drainage and metal mine drainage. Sulfate: 10-30 percent for coal mine drainage and metal mine drainage. Iron: 80-90+ percent for coal mine drainage and metal mine drainage. Aluminum: 90+ percent for coal mine drainage and metal mine drainage. Copper: 80-90+ percent for metal mine drainage. Zinc: 75-90+ percent for metal mine drainage. Cadmium: 75-90+ percent for metal mine drainage. Lead: 80-90+ percent for metal mine drainage.

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EcoBond	EcoBond forms a chemical chain that binds with metal ions forming insoluble metal complexes, reducing bioavailability. It produces a reaction that proceeds at ambient temperatures and does not produce secondary waste streams or gases.	Arsenic, aluminum, cadmium, chromium, lead, mercury, selenium, zinc, radionuclides	Full-scale	Gilt Edge Mine, South Dakota Clear Creek (Gregory Gulch OU), Colorado Golden Sunlight Mines, Inc., near Whitehall, Montana Frontier Hard Chrome, Vancouver, Washington	EcoBond can be deployed as a solid and tilled into acid-generating mine waste, or it can be applied as a liquid and sprayed from a hydroseed cannon to apply the compound to mine pit walls, high angle mine slopes and waste piles, and environmentally sensitive areas, such as riparian zones along streambeds. EcoBond is designed to react with the pyrite within 24 to 48 hours. The pH stabilizes at an environmentally safe level and, as a result, the available Fe+3 in the system decreases.	EcoBond has a 1,000-year simulated durability that has been verified by TCLP leaching parameters.	The EcoBond was not effective at reducing zinc and copper. Since the chemicals are applied with water, the reactions and subsequent effectiveness are limited to the surfaces that can be contacted. This makes treatment at depth or in large stockpiles difficult, since the flow paths in mine waste are complex.	Compared to similar technologies, EcoBond is relatively expensive. The total cost of the implemented technology at the Golden Mines demonstration was $33,934 (2013 USD). The per unit cost for this technology was $10.06 (2013 USD). That figure is based on a unit equaling 2,500 square feet.	Reduced Al by 7 percent, Fe by 26 percent, Mn by 55 percent, Ni by 64 percent, and sulfate by 31 percent. Limited inhibition of copper and zinc. The maximum percent reduction of total metals from the EcoBond treated plot was less than 50 percent.
Electrocoagulation	In electrocoagulation, water is treated using electrolysis with graphite or stainless steel cathodes in conjunction with a metal anode. When a voltage is applied across the electrodes, insoluble precipitates are formed from ions of the metal electrode and selenium, arsenic or other metals present in the water.	Arsenic, copper, lead, zinc, total suspended solids, heavy metals, phosphates	Lab-scale	None	A hybrid electrocoagulation-microfiltration process was tested in the laboratory using industrial wastewater from copper production to remove selenium, arsenic, copper, lead, zinc and cadmium. Water was pre-treated using lime neutralization and sedimentation followed by electrocoagulation. After electrocoagulation, water was filtered using a microfiltration flat sheet ceramic membrane. It then underwent a final lime neutralization step. Anode material selection is dependent on wastewater composition. Electrocoagulation can reduce sludge production significantly compared to other chemical processes such as iron reduction.	Systems are expected to require low maintenance and minimal operator attention. Regular replacement of electrodes is necessary.	High energy consumption. Raises the temperature of the water stream, rendering the direct discharge of the water difficult. The conductivity of the contaminated water must be high. Demonstrated to work more efficiently when lower concentrations of pollutants are present and pH is between 4 and 8.	For some applications, operating costs, including electric power, replacement of electrodes, pump maintenance and labor, can be less than $1.53 (2013 USD) per thousand gallons. Capital cost is significantly less than alternative technologies. A potential cost advantage of the electrocoagulation process is the generation of a lesser amount of sludge. The sludge is generally easier to dewater and may be beneficially recovered.	Not proven for full-scale treatment of mining wastes. Heavy metals in water such as arsenic, cadmium, chromium, lead, nickel and zinc are generally reduced by 95 to 99 percent.

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Electrocoriolysis ELCOR™	Electrocoriolysis is a patented apparatus for separating and removing ionizable components dissolved in water by separating ionizable substances into fractions by the action of electric current and of Coriolis force. Liquid containing ionizable components is continuously fed in the apparatus, and the purified solvent and the solute in a concentrated solution are continuously removed while the liquid is rotated.	Ionizable metals, colloidal solids, particles and inorganic pollutants – besides copper, other heavy metal ions successfully removed from liquid streams by ELCOR™ thus far include Cd+2, Zn+2, Mn+2, Fe+2, and Fe+3	Lab-scale	Unknown	The technology removes metals, colloidal solids and particles, and soluble inorganic pollutants from aqueous media by introducing highly charged polymeric metal hydroxide species.	Unknown	A system capable of being incorporated into fixed or mobile units for ex-situ surface wastewater treatment.	The technology is a continuous process, as opposed to a batch process to achieve cost-effective operation. The energy used is only the energy used in the electrolytic process. For commercial ELCOR™ devices, all sources of energy consumption must be included.	One of the objects of the invention for the dynamic mode is a capacity of water treatment of up to, but not limited to 1.5 million gallons per day (MGD) for a single mobile or transportable unit. Recovery of water of adequate purity for reuse (e.g., irrigation).
Electrodialysis Reversal (EDR)	Electrodialysis (ED) is an electrochemical separation process in which ions are transferred through ion exchange membranes by means of a direct current voltage. EDR is a variation on the ED process, which uses electrode polarity reversal to automatically clean membrane surfaces.	Arsenic, radium, nitrate, dissolved solids	Full-scale	Unspecified mine, South Africa	EDR works the same way as ED, except that the polarity of the DC power is reversed two to four times per hour. When the polarity is reversed, the source water dilute and concentrate compartments are also reversed and so are the chemical reactions at the electrodes. This polarity reversal helps prevent the formation of scale on the membranes.	The ED stack must be disassembled, mechanically cleaned and reassembled at regular intervals. The concentrate waste stream, electrode cleaning flows, and residuals from the pretreatment process will be a part of a typical waste stream flow and will require disposal.	Total dissolved solids (TDS) economical up to 8,000 mg/L, but often run at waters of 1,200 mg/L pH: 2.0 to 11.0 Iron (Fe+2): 0.3 ppm Mn (+2): 0.1 ppm H2S: up to 1 ppm	Unknown	When combined with reverse osmosis at South African mine, the system treats MIW with 5,000 mg/L TDS with high calcium sulfate content down to <40 mg/L TDS.
Evaporation	Evaporation is the vaporization of pure water to concentrate contaminants as a solid or in a brine stream.	Selenium	Pilot-scale	San Joaquin, California	Solar evaporation ponds and enhanced evaporation systems have been examined for selenium treatment. Enhanced evaporation system accelerates evaporation rates by spraying water in the air. The use of mechanical evaporators can produce concentrated brine followed by crystallization, drying and solid waste disposal. Requires minimal energy and no pre-treatment. Mechanical evaporation machines can rapidly increase the evaporation process, with up to 14 times more efficiency than space taken by the same area of pond.	Sediments accumulated during evaporation require disposal. Land-based mechanical evaporation machines require more attention if wind direction varies greatly and if the site is sensitive to spray droplet drift. A pond-based unit requires less operator attention.	Solar evaporation is likely unsuitable for metals removal from MIW due to the prevailing cold climate where operations are typically located. Enhanced evaporation system has not been applied to MIW treatment. Risk of infiltration to ground water (depending on liner type) could occur. May pose a risk to wildlife. An ecological risk assessment should be performed prior to implementation.	The cost of constructing additional storage ponds and the added cost of cleanup and revegetation are often prohibitive. Lower costs because the technology relies on solar radiation for evaporation. Evaporation pond treatment in the San Joaquin valley cost $754 USD per acre-foot of treated water, with $3.3 million/year (2013 USD) for O&M.	Evaporation ponds reduced selenium concentrations by only 25 percent in the San Joaquin Valley.

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Ferrihydrite Adsorption (Iron Co-Precipitation)	Ferrihydrite adsorption is a two-step physical adsorption process in which a ferric salt is added to the water source at proper conditions such that a ferric hydroxide and ferrihydrite precipitate results in concurrent adsorption of selenium on the surface. Also known as iron co-precipitation.	Selenium, arsenic	Full-scale	Kennecott Utah Copper Corporation Garfield Wetlands-Kessler Springs Kennecott North site, Utah	Oxidation of ferrous iron to ferric iron and formation of ferric hydroxide is a common reaction in water that causes iron staining because the ferric hydroxide is insoluble and readily precipitates from water. Oxidation of water soluble ferrous iron to ferric iron is the most common method of removing iron from water. This process is relatively fast at pH values above 6.5 and very rapid at a pH of 8.5 or above. Pretreatment to optimize pH might be required. Flow equalization required as part of the treatment train.	Iron residuals with adsorbed selenium will require thickening and dewatering for disposal as solid waste. Will require toxicity characteristic leaching procedure (TCLP) testing to determine whether or not the sludge should be disposed of as hazardous waste.	Consistent removal to regulatory levels of selenium has not been proven. Potential release of selenium from ferrihydrite residuals. Gravity sedimentation may be required to separate iron solids and adsorbed metals from the water matrix. High operational costs typical of chemical treatment. Ion exchange capacity for selenium can be greatly reduced by competing ions (e.g., sulfates, nitrates).	Cost of a 1 MGD treatment system is estimated at $11.8 million (2013 USD), with an estimated annual O&M cost of about $4.3 million (2013 USD).	At the Garfield Wetlands-Kessler Springs site, water contained 1,950 µg/L selenium, primarily as selenite. Using an iron concentration of 4,800 mg/L, the mean effluent selenium concentration was 90 µg/L. The minimum reported selenium concentration was 35 µg/L. Selenium removal is not proven to less than 5 µg/L.
Fluidized Bed Reactor (FBR)	In a fluidized, or pulsed, bed reactor, contaminated water is passed through a granular solid media at high enough velocities to suspend or fluidize the media, creating a completely mixed reactor configuration for attached biological growth or biofilm.	Selenium, perchlorate, nitrate	Pilot-scale	San Joaquin, California	In this type of reactor, a fluid is passed through a granular solid material at velocities sufficient to suspend or fluidize the solid media. Media types include sand and activated carbon media that are manufactured to exacting specifications for hardness, shape, size, uniformity, density and impurity levels. FBRs allow for shorter residence times for treatment and a smaller overall footprint due to the vertical orientation of the vessels and the efficiency of treatment.	Requires daily cleaning of the influent strainer, tank walls, recycle tank and piping due to biological growth. Envirogen FBR systems are designed to be operated continuously – they do not require cyclical backwash operations.	Presence of excess nitrates necessitates sufficient carbon or energy source, leading to additional biomass. External carbon source may be required. Waste biomass may be hazardous waste.	Total installed cost for a 1 MGD system is estimated at $11.8 million (2013 USD). The annual O&M cost for the same system is estimated at $3.2 million (2013 USD). A recent third-party analysis performed for the North American Mining Council showed that initial capital costs for an FBR system can be one-third or less the cost of a packed bed reactor system designed for similar treatment requirements.	At a pilot test with a flow rate of 1 gpm, total selenium decreased from 520 µg/L to 380 µg/L. Envirogen FBR technology demonstrated the ability to achieve <5 ppb selenium over a 10-month period in treating mining leachate at a U.S.-based coal mining site.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Ion Exchange	Ion exchange is the reversible exchange of contaminant ions from a process stream with more desirable ions of a similar charge adsorbed to solid surfaces known as ion exchange resins. This process provides hardness removal, desalination, alkalinity removal, radioactive waste removal, ammonia removal and metals removal.	Metals, hardness	Pilot-scale	Soudan Park, Minnesota	Ion exchange is generally used as a polishing step to remove low-concentration contaminants, and often requires pre-treatment prior to application. Important considerations include type of resin, the volume and type of regenerant, the backwash water source, backwash quantities, the need for pre-filtration of solids, the column configuration, the need for pH adjustment before and after ion exchange, and the cycle length. Flow equalization/diversion is required as part of the overall system.	Once ion exchange sites on the resin are completely full, the resin must be regenerated in order to be used again. Scale removal may be required to prevent resin fouling. The higher the concentration of TDS in the water, the more frequently the resin will need to be regenerated with caustic soda and rinsed with backwash water.	Pre-filtration may be needed to remove suspended solids that would plug the resin bed. Organics, strong oxidants and high temperatures can degrade the resin. Resins may need to be disposed of if they cannot be regenerated, meaning high disposal costs. High sulfate can result in exhaustion of the resin.	Estimated annual costs for one site, the Soudan Mine discharge (average flow of 86,400 gpd), are about $168,993 (2013 USD).	Generally greater than 90 percent recovery rates, given resin specificity for target constituent and regenerant and back wash requirements. A lab test was performed using process solutions from Kennecott Mining Company containing 0.93 mg/L selenium and 80 mg/L sulfate at pH 4. The resin removed selenium to less than 1 µg/L.
Limestone Pond	Limestone ponds are a passive treatment idea in which a pond is constructed on the upwelling of MIW seep or underground water discharge point. Limestone is placed in the bottom of the pond and the water flows upward through the limestone.	Aluminum, iron, acidity	Pilot-scale	Not Available	Similar to ALDs, this system is recommended for low dissolved oxygen water containing no Fe3+ and Al3+. The advantage of this system is that the operator can observe if limestone coating is occurring because the system is not buried. If coating occurs, the limestone in the pond can be periodically disturbed with a backhoe to uncover the limestone from precipitates or to knock or scrape off the precipitates. If the limestone is exhausted by dissolution and acid neutralization, then more limestone can be added to the pond over the seep.	Replacement of exhausted limestone.	Unknown	Unknown	Unknown

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Nanofiltration Membrane Technology	Nanofiltration is a form of filtration that uses a semi-permeable membrane. The pores are typically much larger than those used in reverse osmosis - close to one nanometer diameter - thus it is not as fine a filtration process as reverse osmosis.	Metals, sulfate	Pilot-scale	Kennecott South, Utah	Similar to reverse osmosis, but operates at one-third the pressure requirement. However, due to larger pore size, it is generally less effective. Requires small space and allows for modular construction. Can offer improved recoveries by rejecting a smaller portion of the salts including selenium, thereby reducing scale potential. Concentrates selenium, reducing the volume for ultimate reduction treatment.	Requires frequent membrane monitoring and maintenance. Membrane life expectancies vary from less than six months to over five years, depending on the quality of the feed solution. Requires treatment and disposal of the residuals.	There are pressure, temperature and pH requirements to meet membrane tolerances.	For a 1 MGD system, total installed cost is estimated at $42.9 million. Annual O&M costs are estimated at $3.2 million (2013 USD). Capital costs for a single-stage filtration unit are quoted at USD $2,392/gpm feed: a plant treating 500 gpm would cost $1.2 million (2013 USD). Additional pre-treatment may be 50 percent of the treatment plant cost, ranging from $359 to $1,196 (2013 USD) per gpm feed. Operating costs are quoted at about $0.60-0.72/1,000 gallons for a nanofiltration unit, with an additional $0.12-0.18/1,000 gallons for additional pretreatment (2013 USD).	Rejection rates: 60 percent for sodium chloride, 80 percent for calcium carbonate, and 98 percent for magnesium sulfate.
Open Limestone Channels (OLCs)	Open limestone channels are the simplest treatment systems, where limestone fragments are added directly to the stream channel semiannually or less frequently. These systems are typically applied when the mine drainage must be conveyed over some distance prior to treatment.	Acidity, manganese, aluminum, iron, copper, lead, zinc, selenium	Full-scale	Many sites in northwest Virginia Brownton, Dola, Florence, Webster and Airport sites Brandy Camp site in Pennsylvania Big Bear Lake near Hazelton in northeastern Preston County, West Virginia Lick Creek in southwestern Indiana McCarty Highwall site in Preston County, West Virginia	An open channel conveying the MIW is lined with high-calcium limestone. The length of the channel and the channel gradient are design factors that can be varied. Optimal performance is attained on slopes exceeding 12 percent, where flow velocities keep precipitates in suspension and where suspended sediments help clean precipitates from limestone surfaces. OLCs can be used alone or in combination with other passive treatment systems. Residence time is critical to OLC performance, yet water velocity must remain high. A settling basin below the channel can be used to slow the water enough to drop out the suspended iron and aluminum hydroxides.	If properly constructed to withstand washout during high flows, OLCs should be nearly maintenance free. Open limestone channels can be useful in abandoned mine reclamation projects where one-time installation costs can be incurred and regular maintenance is not possible.	Limited success where metals are elevated and/or the acidity is also high. The design and operation of the limestone drain require special attention to accommodate the inevitable armoring and coating of the limestone. High-flow velocities to scour settled solids and clean precipitates from the limestone surfaces. Ability to periodically flush the system and clear accumulated precipitates and solids. Burial can be a more significant problem than armoring.	The cost of treatment varied between $32 and $9,303 per ton per year (2013 USD). Most OLCs treat water at or less than $300 per ton per year.	The amount of alkalinity that these systems can usually generate is usually sufficient to raise the pH of the stream to at or near 6. The highest removal rates were with channels on slopes of 45 to 60 percent and for MIW with acidity of 500 to 2,600 mg/L as calcium carbonate. OLCs can be effective as one element of a passive treatment system, but typically are not relied on for stand-alone MIW treatment.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Permeable Reactive Barriers	A permeable reactive barrier (PRB) is a continuous, in situ permeable treatment zone designed to intercept and remediate a contaminant plume. The treatment zone may be created directly using reactive materials such as iron or indirectly using materials designed to stimulate secondary processes, such as by adding carbon substrate and nutrients to enhance microbial activity.	Trace metals, including: chromium, nickel, lead, uranium, technetium, iron, manganese, selenium, copper, cobalt, cadmium, zinc radionuclides, anion contaminants, including: sulfate, nitrate, phosphate, arsenic various methanes, ethanes, ethenes, propanes and aromatics	Full-scale	Durango site, Colorado Nickel Rim, Ontario Monticello Mill Tailings, Utah Tenmile Creek, Montana	Commercial PRBs are currently built as either funnel-and-gate or continuous PRB systems. Both have required some degree of excavation and been limited to relatively shallow depths of 50 to 70 feet or less. The residence time required and the anticipated ground-water velocity through the PRB are used to determine the size of PRB needed to achieve the desired treatment level.	Minimal maintenance is required for PRBs, but performance can decline due to clogging. Depending on several site-specific conditions, PRBs are now expected to last 10 to 30 years before reactivity or hydraulic issues will result in the need for maintenance.	The barrier system may not be a stand-alone technology. The remediation timeframe may require a long treatment period, depending on the size of the contaminated area. Biofouling and mineral precipitation may limit the permeability of the wall system if not managed properly. In both designs, it is necessary to keep the reactive zone permeability equal to or greater than the permeability of the aquifer to avoid diversion of the flowing waters around the reactive zone.	Costs of PRB systems vary depending on site-specific circumstances. The length and especially the depth tend to be the biggest factors that drive the cost of the installation. PRB system installations typically cost more than conventional pump-and-treat technology installations. At the Durango site, treatment costs were about $29.68 (2013 USD) per 1,000 gallons treated. For a PRB installed in Monkstown, Northern Ireland, total treatment costs were $1.4 million (2013USD).	At Monticello, Utah site, influent of 40 µg/L selenium was treated to below detection limits. At the Durango, Colorado site, influent of 359 µg/L selenium was treated to 8 µg/L.
Photoreduction	During photoreduction, ultraviolet light is used to generate electron-hole pairs on the surface of a photocatalyst. Contaminants absorbed to the surface of the photocatalyst undergo redox reactions induced by the electrons and holes created by the exposure to ultraviolet light. The treated species are then desorbed and the surface of the photocatalyst is regenerated.	Selenium	Lab-scale	Unknown	TiO2 has been found to be an effective photocatalyst for the reduction of both selenate and selenite in solution. Using ultraviolet light at wavelengths less than 380 nanometers at a pH of 3.5 in the presence of TiO2 and formic acid will reduce Se(VI) and Se(IV) to Se(0).	Unknown	Unknown	Unknown	Concentrations of 20 µg/L to 40 µg/L of selenate and selenite were tested, with ultraviolet exposure times ranging between two and eight hours producing final effluent concentrations between1 µg/L and 31 µg/L total selenium.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Reverse Osmosis	Reverse osmosis is the pressure-driven separation through a semi-permeable membrane that allows water to pass through while rejecting contaminants.	Metals, sulfate	Full-scale	Kennecott South, Utah	Reverse osmosis is being used at the Kennecott South site as the primary technology for addressing the TDS and sulfate-impacted ground water extracted from the Zone A Sulfate Plume. At the Kennecott South site, reverse osmosis is used with nanofiltration for pre-treatment to avoid reverse osmosis membrane clogging, fouling or damage.	Kennecott site: the manufacturer of the membranes recommended a lifespan of three years, with periodic cleaning cycles. Kennecott's planning and designing of the treatment system and optimizing operational activities around the quality of the feed water has allowed Kennecott to realize about six years of operational life on the membranes.	Requires high operating pressure. Not practical above 10,000 mg/L TDS. Requirements for pre-treatment and chemical addition to reduce scaling/fouling. Reverse osmosis permeate steam will require treatment prior to discharge to receiving waters to meet aquatic toxicity test. Frequent membrane monitoring and maintenance. May require temperature control at low and high temperatures to minimize viscosity effects.	For a 1 MGD system, total installed cost is estimated at $42.9 million (2013 USD). Annual O&M costs are estimated at $3.2 million (2013 USD). Total capital costs for the Bingham Canyon Water Treatment Plant: about $16.1 million (2013 USD). Total yearly O&M costs (40 percent of these costs are labor and 24-hour maintenance expenses) for the Bingham Canyon Water Treatment Plant: about $1.3 million (2013 USD).	Kennecott site Bingham Canyon Water Treatment Plant: has consistently seen permeate production efficiencies in the range of 71 to 72 percent. Demonstrated at full scale to remove selenium to <5 µg/L. Can remove 90 to 98 percent of TDS. A TDS removal efficiency of 98.5 percent was observed during pilot-testing of the membranes tested.
Silica Micro Encapsulation (SME)	SME encapsulates metals in an impervious microscopic silica matrix that prevents the metals from migrating or otherwise adversely affecting human health or the environment. Its physical and chemical components include an initial exothermic reaction and pH adjustment followed by an electrokinetic reaction and metal hydroxyl formation that leads to silica encapsulation.	Aluminum, iron, chromium, copper, lead, mercury, zinc, arsenic, radionuclides	Pilot-scale	Gilt Edge Mine, South Dakota	At an evaluation project at the Gilt Edge Mine, waste rock was treated by building a portable enclosed structure next to the pit. Waste rock was treated in batches before it was loaded into the pit. The treatment facility included the enclosed structure, concrete mixing corral, slurry delivery unit, reagent delivery silos and a water storage tank. SME usually achieves control of contaminants in a single step, without the need for pre-treatment with chemicals or post-treatment flocculation or filtration.	Unknown	Expensive reagents	Total cost of Gilt Edge evaluation was $16.5 million (2013 USD). More than $13 million (2013 USD) of the total cost was spent on reagent.	Gilt Edge Mine waste rock results: Mean iron reduction: 94.82 percent. Mean sulfate reduction mean: 33.18 percent. Mean aluminum reduction: 88.14 percent. Contrary to conventional treatment processes that typically degrade over time, the SME silica matrix continues to strengthen and tighten, further isolating contaminants from the environment.

Technology	Technology Description	Treated Constituents	Scale	Example Sites	Operations	Long-term Maintenance	System Limitations	Costs	Effectiveness
Successive Alkalinity Producing System (SAPS)	Successive alkalinity producing systems (SAPS) combine the use of an ALD and an organic substrate into one system. A SAPS is a pond that contains a combination of limestone and compost overlain by several feet of water. Mine drainage enters at the top of the pond and flows down through the compost, where the drainage gains alkalinity and the oxidation-reduction potential decreases. It then flows into the limestone below. Dissolution of the limestone increases the alkalinity of the water, resulting in precipitation.	Acidity, aluminum, copper, iron, manganese, zinc	Full-scale	Summitville Mine, Colorado Howe Bridge, REM, Schnepp Road, Filson sites, Jefferson County, Pennsylvania	At the Summitville Mine, after the pretreatment of the settling pond, MIW feeds into the SAPS pond for alkalinity treatment and precipitation. Precipitated metals are collected in a subsequent settling pond; discharge from the settling pond was then routed through a polishing channel for final treatment. The total treatment time through the entire treatment system was between 14 and 15 days and about four days through the SAPS ponds. Water with high metal loads can be passed through additional SAPS to reduce high acidity. Iron and aluminum clogging of limestone and pipes can be removed by flushing the system.	Compost removal and replacement. Replenishment of the compost after two to three years may extend the system's effectiveness. Maintenance of bed permeability. Continual monitoring of both influent and effluent water chemistry and metal removal.	With high flows and high metals, more complicated designs may be needed that incorporate treatment cells in a series with increased numbers of settling ponds. More complex systems are costly to build and a larger area is needed. Successful SAPS have used mushroom compost. Other types of organic material have problems with plugging.	The average estimated cost for a SAPS based on a 15-year system life range from $72,439 (2013 USD) for a 5-gpm system to $150,983 (2013 USD) per year for a 100-gpm system. For the 5-gpm system, treatment cost is estimated at $0.03 per gallon of MIW. For the 100-gpm system, the cost is estimated at $0.003 per gallon (2013 USD).	SAPS performance has been inconsistent, but can be effective. Typical observed removal efficiencies: Al – 97 percent Cu — 90 percent Fe — 64 percent Mn — 11 percent Zn — 57 percent
Zero Valent Iron (ZVI)	Zero valent iron (ZVI) can be used to reduce selenium oxyanions to elemental selenium. Ferrous cations can also reduce selenate to selenite and subsequently remove selenite by adsorption to iron hydroxides.	Selenium, arsenic	Pilot-scale	Richmond Hill Mine, South Dakota Dry Valley Mine, Idaho	ZVI acts as a reducing agent in the redox reaction. The iron acts as both a catalyst and an electron donor for the reaction. Systems have typically applied the media in tanks or filter vessels that hold elemental iron. Pre-treatment in the form of pH adjustment may be required. Flow equalization/diversion is required as part of the treatment train.	ZVI media is finite and will require removal, disposal and replacement. Residuals will require TCLP testing to determine whether sludge should be disposed of as hazardous waste.	Requires long contact time. Forms iron oxides and sludge. Passivation and exhaustion of the iron. ZVI treatment is pH and temperature dependent. Due to iron content and reducing environment, aeration followed by clarification is recommended.	For column-based system (using steel wool): total installed cost for a 1 MGD system is estimated at $13.9 million (2013 USD). For stirred-tank based system (using granular ZVI): total installed cost for a 1 MGD system is estimated at $11.8 million (2013 USD). Annual O&M cost is estimated at $3.2 million (2013 USD).	At Richmond Hill Mine, the process is able to remove selenium to concentrations of 12 µg/L to 22 µg/L. Catenary Post pilot: 5 µg/L to 14 µg/L selenium was treated to > 5 µg/L.

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