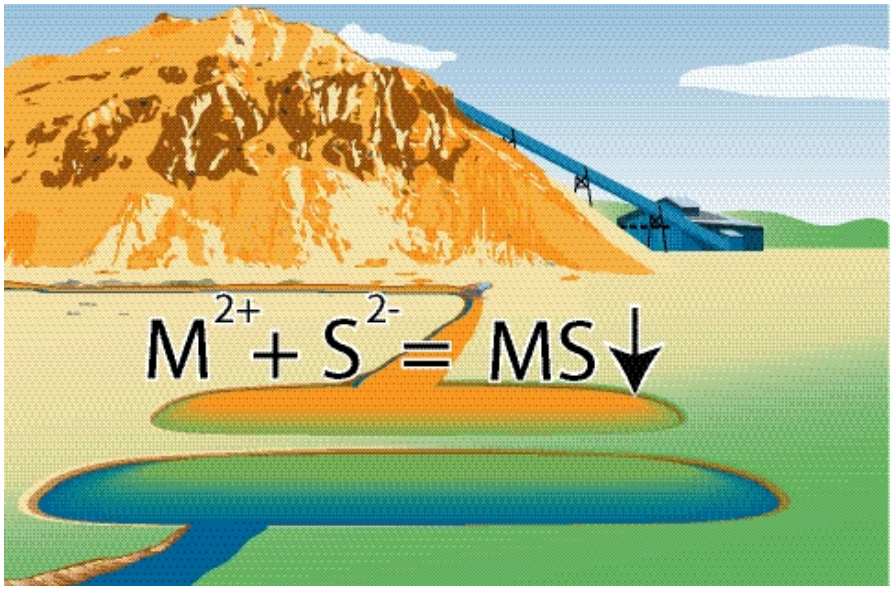
Guidance Document

Biochemical Reactors for Mining-Influenced Water



November 2013

Prepared by
The Interstate Technology & Regulatory Council
Biochemical Reactors for Mining-Influenced Waste Team

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BCR-1

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EXECUTIVE SUMMARY

Mining-influenced water (MIW) includes aqueous wastes generated by ore extraction and processing, as well as mine drainage and tailings runoff. MIW handling, storage, and disposal is a major environmental problem in mining districts throughout the U.S and around the world. During the development of the guidance document entitled *Mining-Waste Treatment Technology Selection* (ITRC 2010), biochemical reactors (BCRs) were assessed in case studies and showed promise as a technology for treatment of these wastes.

A review of BCR applications at mining sites has shown that further investigation and evaluation of BCRs is needed. Specific technology-based guidance regarding BCRs would provide practitioners, consultants, state and federal regulators, stakeholders and responsible parties with a valuable tool to facilitate knowledge transfer, and increase acceptance and use of this innovative technology. The guidance presented here explains BCR technologies used to treat MIW on-site and to improve the ambient water quality in nearby watersheds.

What are BCRs?

BCRs are engineered treatment systems that use an organic substrate to drive microbial and chemical reactions to reduce concentrations of metals, acidity, and sulfate in MIWs. While this chemistry is fairly well understood, advances over the last 20 years have improved the effectiveness, hydraulic permeability, and longevity of these systems. BCRs can be designed to address a wide range of flows, acidity, and metals loading and also can be designed to operate in gravity flow or available powered mode. BCRs may be configured to operate with or without external energy and chemical input, and can often be sustained for months at a time without human intervention.

BCRs can be designed as engineered systems, which use separate tanks or zones for the bioprocesses, chemical reactions, and solids separation. Microbial sorption or precipitation can achieve metals recovery at active mining and mineral processing sites. These systems accommodate reasonably high flow rates and integrate pH adjustment into the water when needed.

BCR systems can also be engineered to incorporate the bioprocesses, chemical reactions, and the bulk of solids separation within an organic substrate. The organic substrate used is generally a mixture of locally available organic materials (such as wood chips or manure) and often contains limestone to provide additional neutralizing capacity and to increase substrate permeability. The optimum substrate design is achieved through extensive preliminary field and laboratory testing using various substrates with site specific MIW. Regardless of the design, BCR systems require monitoring and operation and maintenance to sustain the longevity of the entire system.

BCRs treat MIW by using microorganisms to transform contaminants and to increase pH in the treated water. The most commonly used BCRs for treating MIW are operated anaerobically and are also called “sulfate-reducing” bioreactors (SRBRs, SRBs). The microbial process of sulfate reduction produces sulfide and bicarbonate within the reactor, allowing the target metals such as cadmium, copper, nickel, lead, and zinc in MIW to precipitate as metal sulfides at pH values above 5.0. The bicarbonate promotes an increase in pH and promotes the removal of some metals as car-

bonates such as $FeCO_3$, $MnCO_3$, and $ZnCO_3$ under the appropriate conditions. Conditions may include a specific pH and carbonate concentration.

About This Guidance Document

The ITRC BCR Team developed this guidance as an online document in order to supplement the original web-based Mining Waste Treatment Technology Selection guidance. This new guidance includes both an overview and in-depth analysis of BCR applicability, including the use of a decision tree, testing, design, construction, and monitoring. The decision tree presents the basic factors used to evaluate the efficacy and applicability of BCR treatment for a particular MIW, while several case studies also included in the document present a clear evaluation of diverse site conditions and detailed analysis and evaluation of several different chemical mixtures.

Membership on this team was open to all ITRC members, and the team consisted of a wide variety of state and federal regulators, industry representatives, and stakeholders. Participants with expertise or interest in BCR treatment technology in their states also joined the team and contributed to the development of this guidance.

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1.0 INTRODUCTION

Biochemical reactors (BCRs), like many technologies, can treat mining-influenced water (MIW) using multiple units in sequence. This guidance document refers to BCR treatment systems, but the biochemical reactor is only one unit among a larger sequential treatment system. A BCR unit, in this guidance, is:

> …an engineered treatment system that uses an organic substrate (electron donor) to drive microbial and chemical reactions to reduce concentrations of metals, acidity, and sulfate in mining-influenced water (MIW).

This broad definition encompasses many of the technologies identified in Table 1-1, including successive alkalinity producing systems (SAPS)/reducing alkalinity producing systems (RAPS), permeable reactive barriers (PRBs), anaerobic constructed wetlands, sulfate reducing bioreactors (BCR), and aerobic constructed wetlands. However, this document focuses only on the BCR technology based on specific design considerations.

BCRs can be designed to treat MIW at abandoned mine sites as well and active mining operations. A BCR can be suitable for remote sites and can potentially operate with minimal operation and maintenance for long periods of time. However, optimal performance of a BCR requires long-term monitoring and maintenance. Although BCRs can have various applications, this guidance emphasizes solid substrate BCRs for remediating MIW. Liquid substrates may be a design consideration in some instances. Additionally, MIW from a mine pool or pit lake could be pumped to an ex situ BCR in certain situations or a mine pool could be treated in situ using the same biochemical treatment processes. Like any treatment technology, proper influent characterization, testing, siting, design, monitoring and maintenance is required to minimize the risk of failure.

The advantages of using a BCR to treat MIW include:

- treatment for an wide variety of constituents
- low operational and maintenance (O&M) requirements
- low initial cost of material for construction, especially when using local material for substrate
- uses basic construction practices
- can be implemented in remote areas, can be self-sustaining for months without human intervention.

Disadvantages of using a BCR to treat MIW include:

- rarely a stand-alone treatment, often requiring pre- and posttreatment
- may not consistently meet water quality criteria
- uncertainty of the operation and maintenance costs,
- space may limit an effective design of a BCR

These advantages and limitations are discussed in more detail in this guidance and a decision tree is included to assist in determining the applicability of a BCR for particular sites. Limitations and possible design alternatives (pre- and posttreatment) are also included.

1.1 Objective of This Guidance

The purpose of this guidance is to help you to understand and apply the physical, chemical, and biological mechanisms that occur in a BCR. This guidance defines the key questions and decision points for assessing the suitability of BCR treatment based on relevant characteristics of the MIW and the site. When a BCR is suitable, this guidance explains the application, testing, construction, operation, monitoring, and maintenance of these treatment units. Since BCRs are seldom stand-alone treatment systems, pre- and posttreatment options are listed and briefly described (Section 2.5). Additional information on these pre- and posttreatment systems can be found in ITRC MW-1 2010.

Although it is not a design manual, this guidance offers a decision tree that describes the features the user should consider when assessing the use of a BCR. The guidance outlines general design guidelines, testing protocols, cost elements, construction (scale-up considerations), troubleshooting, and maintenance. Case studies highlight the applications and challenges of BCRs.

This guidance can assist tribal, state, and federal regulators (including permit writers, compliance staff, and technical support staff), federal land managers, practitioners and consultants, industry (including site owners), public and private researchers, and the international mining industry. This guidance is also a valuable resource for community and public stakeholders (such as watershed groups), environmental protection and land resource nongovernment organizations, and educators.

1.2 Overview of BCR Treatment for Mining-Influenced Water

It has been over thirty years since the pioneering work of a group of researchers at Wright State University documented water quality improvements in a natural sphagnum bog in Ohio that received low pH, metal laden water (Huntsman et al. 1978). After a second group at West Virginia University found similar results at the Tub Run Bog (Lang et al. 1982), researchers and practitioners focused on developing the promising technology of using constructed wetlands (ITRC WTLND-1 2003) to treat acid mine drainage (AMD) or acid rock drainage (ARD), both of which could be considered an acidic variant of MIW. In 1993, Wildeman, Brodie, and Gusek published *Wetlands Design for Mining Operations*, which focused on the development of the treatment method now called "biochemical reactors." As the technology has developed and broadened, the term "passive treatment" was coined to describe additional, related systems such as those listed in Table 1-1.

Table 1-1.
Overview of MIW passive treatment systems.

Source: Adapted from Doshi (2006).

Technology	Description	Primary Functions
Anoxic limestone drain (ALD) (http://www.itrcweb.org/miningwaste-guidance/to_anoxic.htm), Open Limestone Channel (OLC), and Limestone Bed	Limestone channel with minimal air contact	Raise pH due to carbonate dissolution, limit metal oxide formation
Successive/Reducing Alkalinity Producing System (SAP)/(RAP), Vertical Flow Pond (VFP)	Limestone and organic mixture, usually downflow	Raise pH due to carbonate dissolution and sulfate reduction
Aerobic wetland (http://www.itrcweb.org/miningwaste-guidance/to_const_treat.htm)	Shallow substrate layer with plants and often open water; usually horizontal flow	Remove metals by metal oxide formation and some removal by adsorption and sulfide formation
Anaerobic wetland (http://www.itrcweb.org/miningwaste-guidance/to_const_treat.htm)	Organic substrate bed with subsurface flow; plants often present, usually down flow	Remove metals by metal sulfide formation and some removal by adsorption
Biochemical Reactor (BCR) also called Sulfate Reducing Bioreactor	Organic substrate bed with subsurface flow, usually down flow	Remove metals by metal sulfide formation and some removal by adsorption
Permeable Reactive Barrier (http://www.itrcweb.org/miningwaste-guidance/to_permeable.htm)	High hydraulic conductivity media (sand, often) with reactive component, designed to capture groundwater in situ	Reactive component based on function: • alkalinity addition = limestone, • metal co-precipitation = zero valent iron, • metal sulfide formation = organic substrate

Aerobic/anaerobic wetlands and some permeable reactive barriers used to treat MIW share many of the same technical features as BCRs and are now considered to be included under the broad term "biochemical reactor."

Because no single treatment unit type works in every situation or with every MIW geochemistry, a BCR treatment system often includes pre- or posttreatment units. For example, manganese in MIW is typically unaffected in a BCR unit; an aerobic polishing unit or similar post-BCR treatment unit is required to address manganese. Pretreatment may be required to reduce acidity or reduce metals loading, specifically for iron and aluminum. An example (Nairn et al. 2010) of a multi-unit treatment system containing a BCR is the Tar Creek unit shown in Figure 1-1.

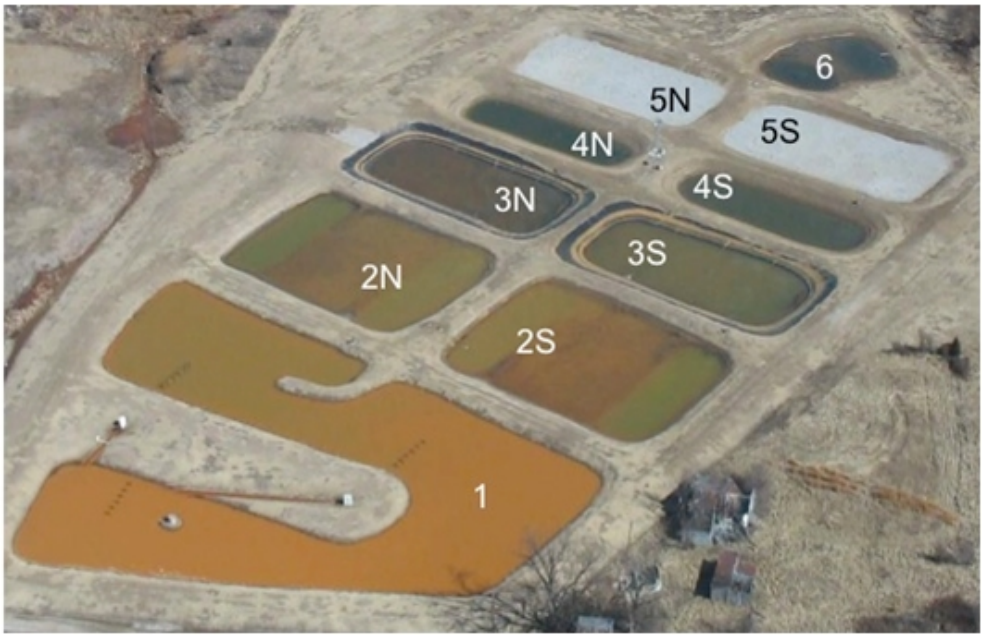


Figure 1-1. Tar Creek passive treatment system in 2009.

Source: Image used with permission from Nairn et al. 2010 (see Appendix B.2, Mayer Ranch Case Study).

This treatment system uses a sequence of five treatment unit types in series, with paired, parallel units for four of those steps:

1. The circumneutral (pH 6) MIW flows into an oxidation pond shown at the bottom of the image.
2. The water flows through parallel surface-flow aerobic wetlands (2N and 2S).
3. Water flows through a pair of parallel downflow BCRs (3N and 3S).
4. The water is aerated by surface flow and bubblers powered by solar and wind energy, through shallow ponds (4N and 4S).
5. The final unit is a pair of horizontal-flow limestone beds (5N and 5S).
6. The treated effluent then is combined into a pond/wetland (6) for polishing.

A BCR treatment system should function for years without organic substrate refurbishing or replacement and in most cases be able to function without electrical power. Beining and Ott (1997) described a volunteer passive system exhibiting BCR characteristics outside of an abandoned lead-zinc mine in Ireland that has apparently been functioning unattended for over 120 years. Ideally, these treatment systems should be designed to last for decades.

1.3 Chemical Reactions and Equilibrium in BCRs

BCR systems consist often of both chemical and biochemical reactions. The overall biogeochemistry can be complex, involving multiple reactions. This section discusses the biogeochemistry and the use of Eh-pH diagrams (Stumm and Morgan 1996), which can help guide the BCR design. Eh-pH diagrams define possible stable (equilibrium) phases of an aqueous electrochemical system. Like phase diagrams, Eh-pH diagrams do not allow for reaction rate or kinetic effects. The

minerals defined within the bound zones are thermodynamically stable and, once the mineral is formed, remain indefinitely in this composition unless the geochemistry of the system changes.

Each site and BCR has its own Eh-pH diagram. Since each site can be complex, the objective here is to provide a general overview. Each site will have its own unique combination of chemical species and microbiological reactions. Site specific treatment options can affect Eh-pH diagrams as well. For instance, the site water is often acidic and calcium carbonate ($CaCO_3$) is used in various unit configurations to raise and maintain the pH range to improve treatment efficiency in a BCR.

In addition to the pH adjustment by $CaCO_3$ addition, a carbon source is added to the BCR and often consists of complex organics such as wood chips or manure. This mixture slowly degrades over many years, providing bacteria with the necessary substrate and food to support the biochemical reaction required to precipitate metals. Microorganisms use the organic carbon and available electron acceptors, resulting in conditions that promote metal sulfide precipitation. The most energetically available electron acceptors are consumed first (such as oxygen, nitrate, nitrite, ferric iron) generating reducing conditions suitable for biological sulfate reduction to sulfide.

1.3.1 Chemistry of a BCR

Certain metals are also reduced in these conditions. Simultaneously, alkalinity, added to or generated in the BCR, produces pH conditions that drive formation of metal sulfide solids. These solids are then trapped in the solid substrate of the BCR or captured in sedimentation cells downstream of the BCR thereby removing the metals from the MIW (Section 4.3). Many metals precipitate as metal sulfides while others simply change to lower valences that result in insoluble minerals. Figure 1-2 illustrates metal species that transform due to a reduction in valance due to biochemical reactions.

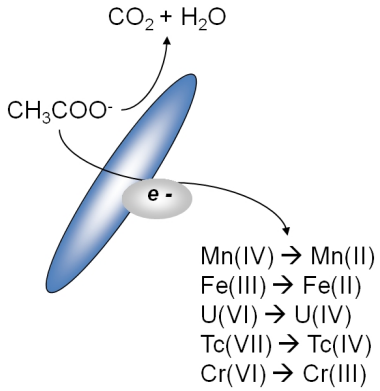


Figure 1-2. Biochemical reduction of some important metals (CH_3OO - acetate).

Source: Dr. Eric Nuttall, University of New Mexico Professor Emeritus.

Often, and particularly in the case of uranium reduction, (see Appendix B.12, Uranium Remediation Case Study) the reduction in valence results in an insoluble species. See the biochemical reaction for uranium illustrated in Figure 1-3.

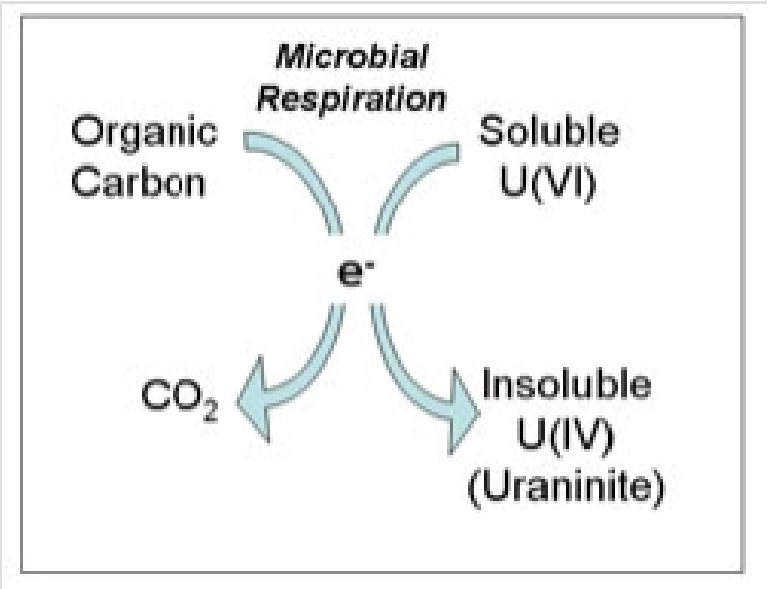


Figure 1-3. Biochemical reduction of uranium of soluble U^{+6} to insoluble U^{+4}

Source: Dr. Eric Nuttall University of New Mexico Professor Emeritus.

The reactions of interest in a BCR are determined by the interactions of the organic substrate, the microbial consortium, and the water flowing through that substrate. Because BCRs may be applied to MIWs ranging widely in pH and containing various metals, the explanation could be exhaustive; instead, only the most commonly encountered mechanisms for various contaminants are presented here. Chemical equilibrium software such as MINTEQA2 (http://www.epa.gov/exposure-assessment-models/minteqa2), PHREEQC (http://wwwbrr.cr.usgs.gov/projects/GWC_coupled/phreeqc), or Geochemist's Workbench (http://crustal.usgs.gov/projects/aqueous_geochemistry/geochemists_workbench.html), among others can be used to determine the likely species for a specific water composition as it is subjected to changes in pH and redox potential (Eh), as shown in Figure 1-4.

Ferric iron soluble at pH < 3.0

Ferrous iron soluble at pH < 3.0 in presence of O_2, but soluble up to pH 6 in absence of O_2

Therefore, it is possible to have anaerobic ARD with pH 6 (little proton acidity), but significant dissolved ferrous iron (mineral acidity)

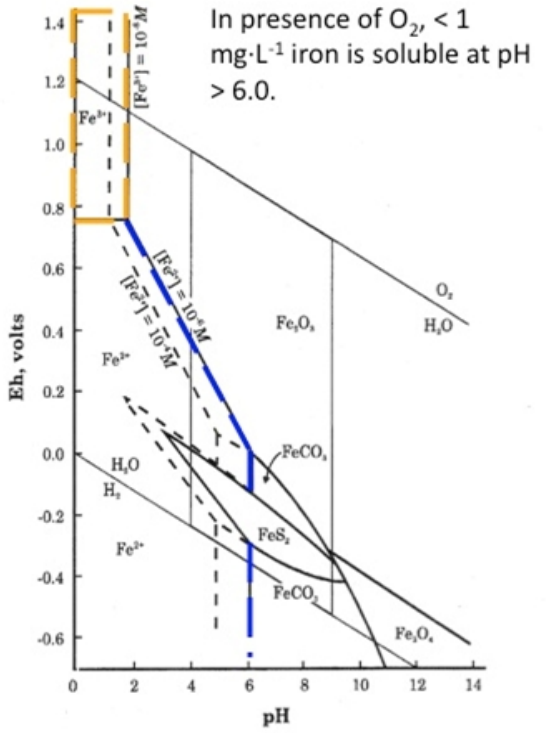


Figure 1-4. Eh-pH diagram showing stability fields of common iron minerals (ferric iron, Fe^{3+}, and ferrous iron, Fe^{2+}]. Total activity of dissolved carbonate 1M and dissolved oxygen 10^{-4} M. Solid boundaries represent 10^{-4}M iron: dashed lines 10^{-4}M.

Source: Robert Thomas, CH2M Hill.

Combining iron and uranium elements onto one Eh-pH diagram (see Figure 1-5) shows the formation of in soluble U^{+4} and at a lower Eh than the formation of FeS_2. These diagrams show the change in chemical minerals as the system become more anaerobic and as the pH shifts from low to high. As the system becomes more anaerobic, the Eh will decrease.

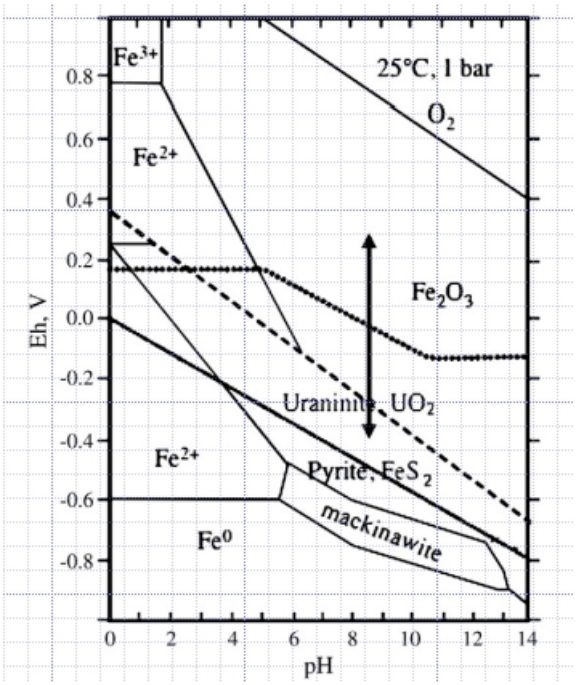


Figure 1-5. Eh-pH diagram for the combined case of iron and uranium.

Source: Dr. Eric Nuttall, Dankritz Case Study.

1.3.2 Microbiology of a BCR

The microbiological community in sulfate-reducing treatment systems includes several major classifications of microorganisms with different types of metabolic activity (Figure 1-6). These microorganisms include sulfate-reducing bacteria (SRB), cellulose degraders, and fermenters (Hiibel et al. 2008, Logan 2005). SRB depend on and are influenced by the metabolic activity of these other groups of microorganisms (Neculita et al. 2007).

SRB respire anaerobically using sulfate as a terminal electron acceptor and a simple organic carbon source as an electron donor. Different genera of SRB use different carbon sources (for example; acetic acid, hydrogen, simple sugars), and a variety of genera are found in a typical BCR (Prieto et al. 2008). SRB are found within a number of taxonomic groups including *Deltaproteobacteria* (for instance, *Desulfovibrio*, *Desulfomicrobium*, and *Desulfobacter*) and spore-forming bacteria such as *Desulfomaculum* (Doshi 2006).

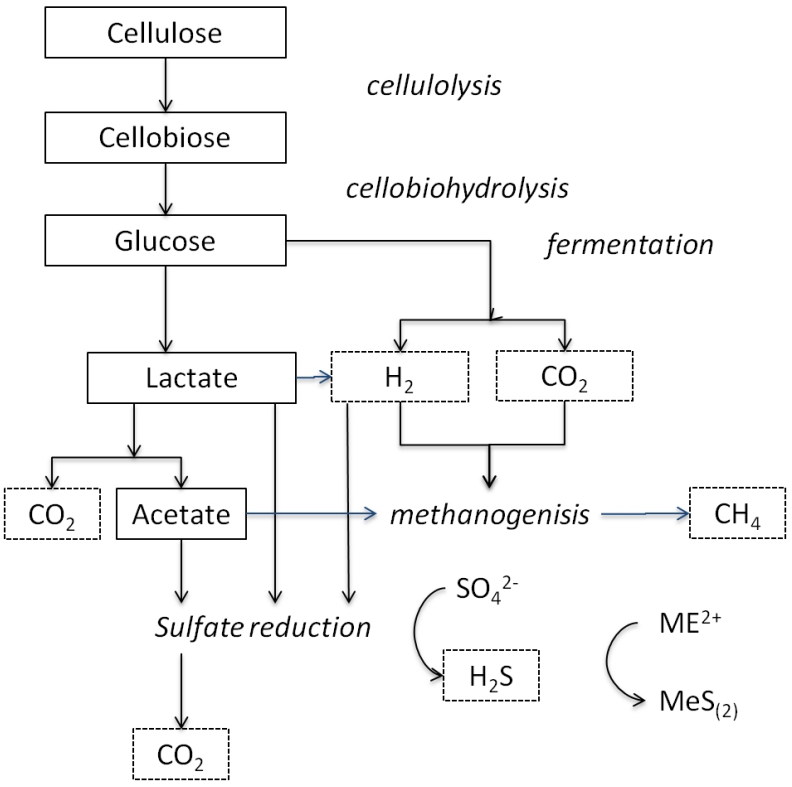


Figure 1-6. Microbial processes involved in BCRs.

Source: Logan et al. 2005.

Temperature affects the growth of SRB. *Deltaproteobacteria* are gram-negative mesophilic bacteria (optimal growth temperature range 20–40°C) with a variety of shapes and physiologic traits. Most bacterial species prefer a similar temperature range to *Deltaproteobacteria* although some can withstand higher temperature environments. Thermophilic SRBs thrive at the high temperatures found in environments such as geothermal vents; however, these bacteria are usually not present in BCR treatment systems.

Cellulose degraders are necessary for bioreactors because they degrade the substrate, which is typically a complex carbohydrate, into simple carbon compounds. SRB cannot degrade cellulose and depend on the cellulose degraders to provide the carbon source (Neculita et al. 2007). Cellulose degradation can be either aerobic or anaerobic, with fermentative anaerobes likely to predominate in a sulfate-reducing treatment system. Examples of bacteria that are cellulose degraders include members of the bacterial genera *Clostridium*, *Ruminococcus*, and *Caldicellulosiruptor*. A wide variety of fungi also degrade cellulose, including members of the groups *Ascomycetes*, *Basidiomycetes*, and *Deuteromycetes*.

Fermenters are important in sulfate reducing treatment systems because they degrade amino acids, sugars, and fatty acids into simpler organic compounds (for example propionic acid and alcohols) that can also be used as carbon sources by SRB. Sometimes called "fermentative acidogens", these bacteria perform both fermentation and acidogenesis. Fermentation is an anaerobic metabolic

process that does not depend on external electron donors or acceptors (unlike respiration) and in which the carbon source is both reduced, and a portion is oxidized, to produce the energy that the unit must have to thrive (Madigan et al. 2012). Acidogenesis is a stage in anaerobic digestion in which molecules such as sugars and amino acids are further broken down to simple fatty acids and alcohols (Figure 1-7).

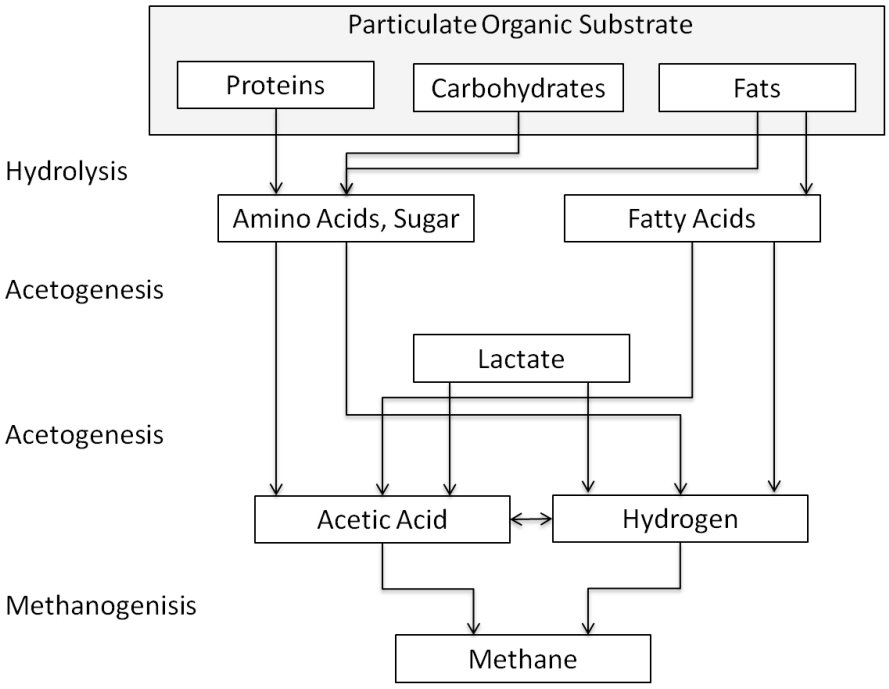


Figure 1-7. Anaerobic digestion process.

Source: Waste To Energy Research and Technology, www.wtert.eu.

Hydrogen ion (H^+) and molecular hydrogen are produced by a number of different fermentative bacteria in a bioreactor. Examples of hydrogen-producing fermentative reactions include the degradation of lactate to produce ethanol, carbon dioxide, and hydrogen ion by Lactobacillis and the degradation of glucose to butyrate, hydrogen gas, carbon dioxide, and hydrogen ion by *Clostridium* butyricum. Biological activity by fermenters has been found to significantly decrease as SRB activity increases in the initial stages of bench-scale passive treatment system operation (Logan et al. 2005).

Some microorganisms, including methanogens (a group of anaerobic *Archaea*) and a few genera of SRB, use molecular hydrogen as an electron donor. Studies have shown that SRB out-compete methanogens for available hydrogen, and methanogen activity is not expected to be high in a mature BCR (Logan et al. 2005).

If the BCR becomes exposed to oxygen, bacteria of the genus *Thiobacillus* (reclassified to *Acidithiobacillus*, Kelley and Wood 2000) can become more predominant (Hiibel et al. 2008). *Thiobacilli* are common bacteria in untreated MIW, are acidophilic, and aerobically oxidize ferrous iron to ferric iron.

Additionally microorganisms can be used to remove dissolved metals or other inorganic contaminants (such as uranium, tellurium, selenium, and chromium oxides). Microbes have the ability to incorporate metals into their structure. Contaminants are removed from the MIW and sorbed on the biomass within the bioreactor. The immobilized biomass is capable of ion exchange or chemical reduction. Microbe selection should be based on the organisms' nutritional requirements available in the bioreactor (for example, sources of carbon, nitrogen, sulfur, and phosphorus), light requirements, and potential pathogenicity. The particular genus and species of the organisms are largely irrelevant, although organisms isolated from the site (or one similarly contaminated) would be most likely to carry out the desired reduction.

1.3.3 Reaction Zones in a BCR

Although different flow configurations are possible, a downflow BCR typically consist of a free-water surface zone, and three separate reactive zones, as shown in Figure 1-8. Listed below, these zones are characterized by decreasing redox potential (related to changes in electron acceptance):

- free water interface
- oxidative zone
- transitional (anoxic to slightly anaerobic) zone (organic depleted zone–active microbial reaction zone)
- sulfide (anaerobic, highly reduced) zone (active microbial reaction zone)

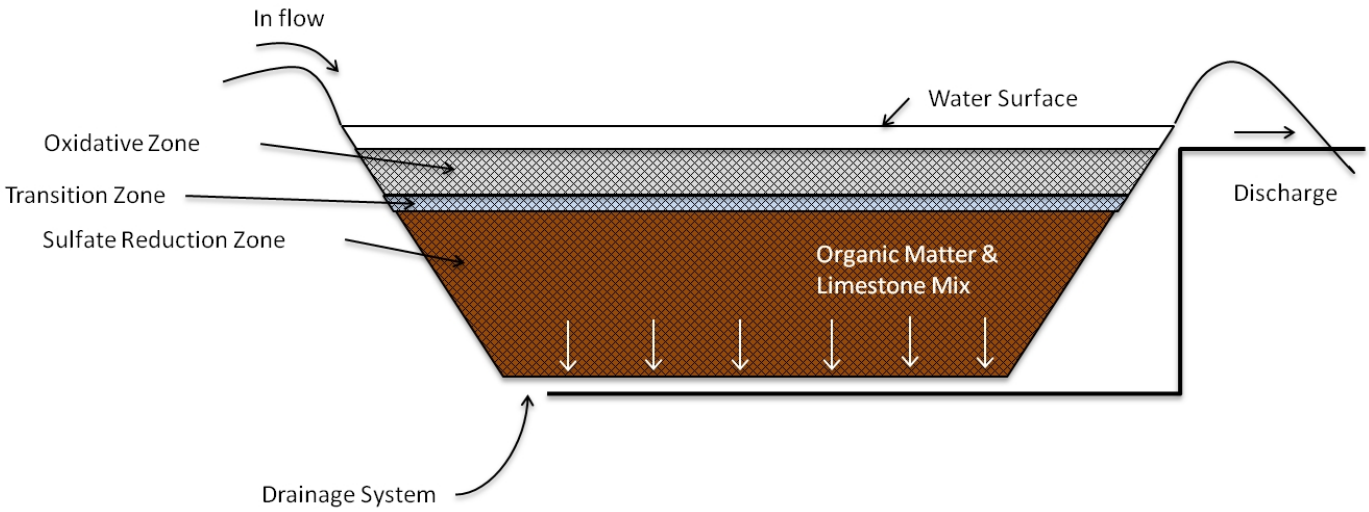


Figure 1-8. BCR reaction zones, schematic view in a typical downflow bioreactor (not to scale).

The physical, chemical, and biological mechanisms at work in each zone vary. These geochemical and biogeochemical reactions typically occur in an oxidizing environment in contact with air. When the percolating solutions encounter reducing conditions, sulfide minerals form in a steadily advancing enriched zone. At start-up, the oxide zone is relatively thin, as is the underlying transition zone. The sulfide zone extends from the bottom of the transition zone to the drainage

collection layer that typically consists of crushed gravel and perforated pipes. This section first presents the general chemical and biochemical mechanisms occurring in a BCR. Then, these mechanisms are described as they occur in the reaction zones for a downflow BCR treating acidic and high iron and aluminum MIW. Portions of the chemistry identified in Table 1-2 (and explained further in Appendix A) are observed in each of the three zones of a BCR. A summary of zones and major reactions is also shown in Table 1-2.

Table 1-2. BCR zones and chemistry of interest

Zone	Description	Simplified Chemistry
Free Water	Surface water adjacent to media	$4\ Fe^{2+} + O^2 + 2H_2O \rightarrow 4\ Fe^{3+} + 4\ OH^-$ $Al^{3+} + 3\ H_2O \rightarrow Al(OH)_3 + 3H^+$
Oxide	Media in contact with oxygenated water including water near roots	$CH_2O \rightarrow H_2O + C$ $CaCO_3(s) + 2\ H^+ \rightarrow Ca^{2+} + H_2CO_3(g)$ $Cellulose + O_2 \rightarrow CO_2 + H_2O$ $Fe^{3+} + 3\ H_2O \rightarrow Fe(OH)_3(s) + 3\ H^+$ $6\ HCO^{3-} + 6\ H^+ \rightarrow C_6H_{12}O_6 + 6\ O_2$
Transitional	Dissolved oxygen (DO) falling to zero in media, not highly reduced	$e^- + Fe^{3+} \rightarrow Fe^{2+}$ $Fe^{2+} + CO_3^{2-} \rightarrow FeCO_3(s)$ $Fe^{2+} + CaCO_3(s) \rightarrow FeCO_3(s) + Ca^{2+}$ $Mg^{+2} + CaCO_3 \rightarrow MgCO_3 + Ca^{+2}$ $2NO_3^- + 10e^- + 10H^+ \rightarrow N_2 + 6H_2O$ $Mn^{2+} + CaCO_3 + CO_3^{2-} \rightarrow MnCO_3(s) + Ca^{+2}$
Sulfide	Anaerobic, highly reduced	Cellulose → Organic acid $Organic\ acid + SO_4^{2-} \rightarrow CO_2 + H_2O + S^{2-}$ $Metal^{2+} + S^{2-} \rightarrow Metal\ Sulfide\ (s)$

1.3.3.1 Free Water Surface Zone

A free water surface zone should evenly distribute MIW over the surface of a downflow BCR. Either a two to six inch layer of shallow standing water on the top of the BCR or, in place of standing water, gravel or plastic infiltration structures can be used to establish even distribution of MIW. Perforated pipes are not recommended for use in distributing MIW within the BCR because they are prone to plugging and causing short-circuiting within the BCR due to precipitation of ferric iron and aluminum oxyhydroxides (Gusek 2004).

MIW flowing into a BCR may have a pH less than 4.0 and could contain ferric iron (Fe^{+3}) and aluminum (Al^{+3}) in solution. If ferrous iron is present, it will likely oxidize to ferric iron:

1. $4Fe^{2+} + O_2 + 2H_20 \rightarrow 4Fe^{3+} + 4OH^-$

As an example this reaction consumes hydrogen ions, so the pH tends to temporarily rise. If any aluminum is present in the mine water, a minor amount of aluminum hydroxide may precipitate:

2. $Al^{3+} + 3H_2O \rightarrow Al(OH)_3$ [Gibbsite - amorphous] $+ 3H^+$

This reaction was observed in the lime-free aerobic wetland at the Wheal Jane site in Cornwall, England (see Appendix B.9).

1.3.3.2 Oxide Zone

In the oxide zone, acidic MIW attacks the cellulose in the organic substrate in a dehydration reaction in regions where limestone (a typical BCR substrate component) has been consumed. This may result in a small amount of hydrogen ion consumption which raises the local pH, allowing iron oxy-hydroxide to precipitate. Since most BCR is mixed with limestone (containing calcium carbonate, $CaCO_3$) to buffer acidity, the following reaction is likely to dominate in the oxide zone or the transition zone described below, which, depending on pH, produces bicarbonate (not shown) and carbonic acid.

This hydrogen ion consumption raises the local pH, allowing iron oxy-hydroxide to precipitate.

3. $CH_2O \rightarrow H_2O + C$
4. $CO_2{:}CaCO_3 + 2H^+ \rightarrow Ca^{+2} + H_2CO_3$ ($H_2CO_3D\ H_2O + CO_2$)

The presence of oxygen also allows cellulose-degrading bacteria to breakdown the organic matter (see Figure 1-9).

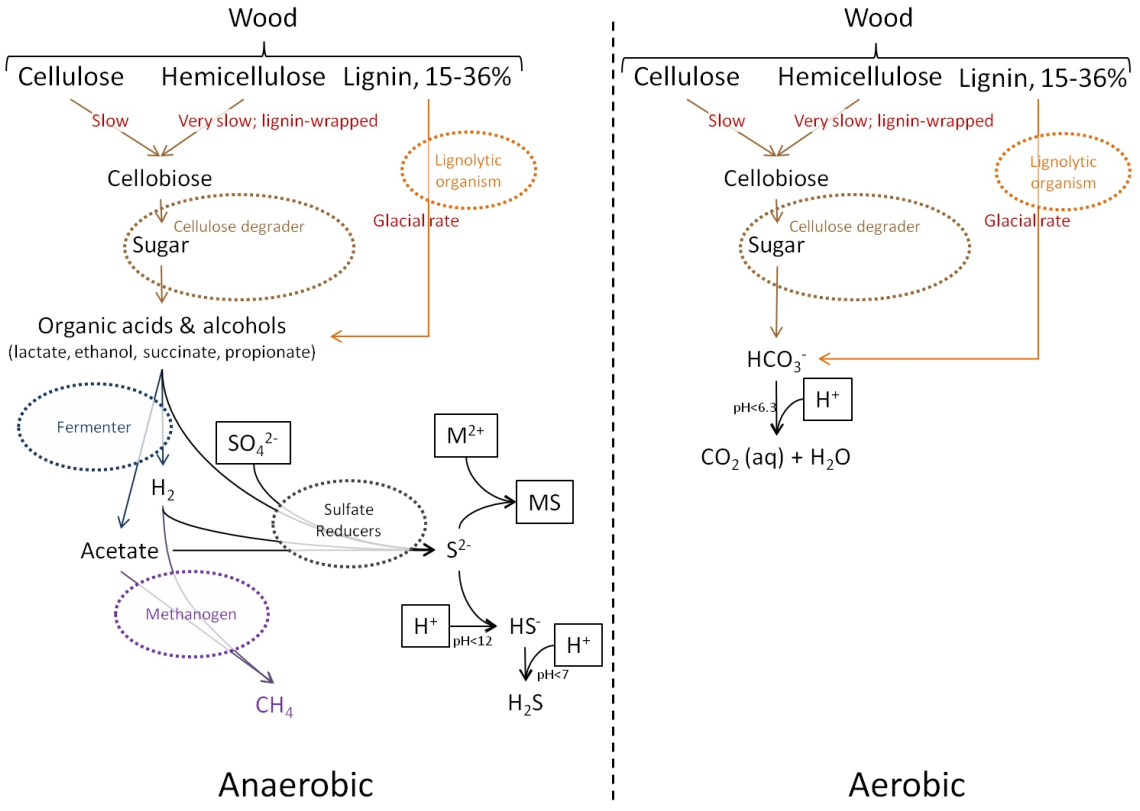


Figure 1-9. Conceptual food chain in a BCR using woody substrate. CO_2 and H_2O omitted.

Source: Adapted from Seyler et al. (2003).

With the elevation of pH, iron oxyhydroxides should precipitate in accordance with the following equation:

5. $Fe^{+3} + 3\ H_2O \rightarrow Fe(OH)_3(s) + 3\ H^+$

As the organic matter and limestone are consumed, the oxy-hydroxide zone expands downward. If the MIW is very acidic, the solid oxyhydroxides might actually be re-dissolved and re-deposited at the bottom of the oxidizing zone or in the transitional zone.

Plants are typically not part of a BCR design but can colonize the surface of the BCR and affect its chemistry. Oxygen diffuses from leaves to roots, creating a zone of oxidation surrounding the root surface known as an oxidized rhizosphere (ITRC Phyto-3 2009). In addition, photosynthesis reactions drive ion uptake by roots, which can further increase the pH in the root zone because the use of carbon dioxide affects the carbonate equilibrium (Wildeman et al. 1993):

6. $6\ HCO_3^-(aq) + 6\ H^+ \rightarrow C_6H_{12}O_6 + 6\ O_2$

During the active growing season, the effects of oxygen diffusion and metabolic processes of the plant and fauna within the rhizosphere may dominate the oxidizing zone. These processes may convert narrow portions of the transitional and sulfide zones discussed below to oxide zone, liberating metals that had been precipitated there. At night plants use the sugar formed during photosynthesis, resulting in increases in CO_2 and possible diurnal pH swings in systems dominated by plant activity. When plants go dormant in the winter, lower portions of the oxide zone may revert to transitional and sulfide zone reactions over a period of weeks. Because of these effects, plant colonization on the top of a BCR should be carefully considered when designing or managing these systems.

1.3.3.3 Transitional Zone

Stripping of dissolved oxygen occurs in the transitional zone, primarily through biological decay of organic matter. Ferric iron (Fe^{3+}) is reduced to ferrous iron (Fe^{+4}) (Thomas et al. 2002):

7. $Fe^{+3} + e^- \rightarrow Fe^{+2}$

If a BCR is being overloaded, test evidence suggests the formation of the mineral siderite in accordance with the following reaction, where iron replaces calcium in the limestone:

8. $Fe^{+2} + CaCO_3 \rightarrow FeCO_3$ [siderite] $+ Ca^{+2}$

A similar phenomenon is possible if excess magnesium is present in the mine water:

9. $Mg^{+2} + CaCO_3 \rightarrow MgCO_3$ [dolomite] $+ Ca^{+2}$

A more common consequence of BCR overloading is the temporary removal of manganese. As discussed in Section 6.0, manganese removal in a BCR does not occur because of the reducing geochemistry. However, during overloading conditions, manganese removal often occurs. The mineral rhodochrosite is likely being formed in accordance with the following reaction:

10. $Mn^{+2} + CaCO_3 \rightarrow MnCO_3$ [rhodochrosite] $+ Ca^{+2}$

These three carbonate mineral phases are subject to subsequent dissolution from acidic BCR influent or, as in the case of the rhodochrosite, dissolution when overloading conditions abate, and the geochemistry surrounding the rhodochrosite grains becomes reducing.

The formation of iron, magnesium, or manganese carbonates is reversible, which can affect BCR effluent concentrations. When these mineral phases re-dissolve, the concentrations of manganese, iron, or magnesium in the BCR effluent may temporarily be higher than the respective concentrations in the influent. This condition may persist for weeks, until the carbonate minerals are consumed (Reisman et al. 2003). Some of the increase in concentrations may also be due to freeing of adsorbed (physically bound) compounds from the substrate.

11. $2\,NO_3^- + 10e^- + 12H^+ \rightarrow N_2 + 6H_2O$

Nitrate is reduced to nitrogen gas in a BCR as the dissolved oxygen decreases.

1.3.3.4 Sulfide Zone

In the sulfide zone, the SRB produce sulfide and bicarbonate (which can also raise the pH of the BCR effluent) in accordance with the following approximate reactions shown as equations (Wildeman et al. 2005):

11. $SO_4^{-2} + 2CH_2O \rightarrow HS^- + 2\,HCO_3^- + H^+$

The dissolved sulfide ion can precipitate metals as sulfides, essentially reversing the reactions that occurred to produce MIW. For example, the following reaction occurs for dissolved zinc, typically forming amorphous zinc sulfide (ZnS):

12. $Zn^{+2} + HS^- \rightarrow ZnS\,(s) + H^+$

These sulfides are thought to precipitate in the void spaces between the particles of limestone and organic matter in the substrate, but can form and infill the cells of the cellulose as shown in Figure 1-10 (Thomas 2002).

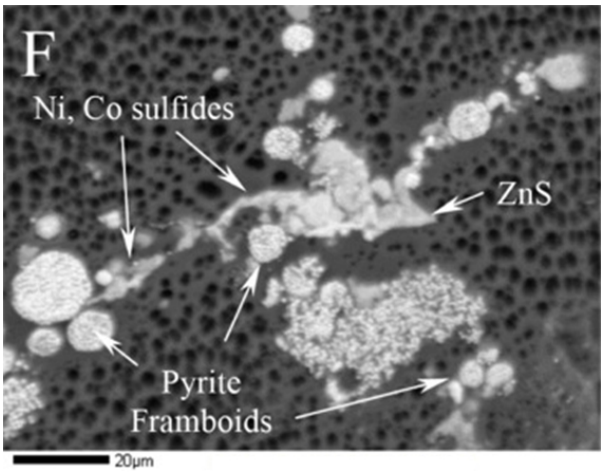


Figure 1-10. Cellulose cell infilling by metal sulfide precipitates.

Source: Thomas and Romanek (2002).

While aluminum does not form a sulfide, it is commonly removed in BCR environments. In one case, instead of aluminum hydroxide, $Al(OH)_3$, typically formed in SAPS, zones of insoluble aluminum hydroxyl-sulfate precipitates occurred in portions of a limestone buffered organic substrate

(LBOS)-filled units, similar to BCRs. The following reactions may apply (Thomas and Romanek 2002):

13. $Al^{3+} + H_2O + SO_4^{2-} \rightarrow Al(SO_4)(OH) + H^+$
14. $3Al^{3+} + K^+ + 6H_2O + 2SO_4^{2-} \rightarrow KAl_3(OH)_6(SO_4)_2$ [Alunite] $+ 6H^+$
15. $6Ca^{2+} + 2Al^{3+} + 38H_2O + 3SO_4^{2-} \rightarrow Ca_6Al_2(SO_4)_3(OH)_{12}{\cdot}26H_2O$ [Ettringite] $+ 12H^+$

The formation of such aluminum hydroxyl-sulfates may be related to a replacement reaction with limestone as suggested by the limestone fragment ghost observed by Thomas and Romanek (2002) and shown in Figure 1-11.

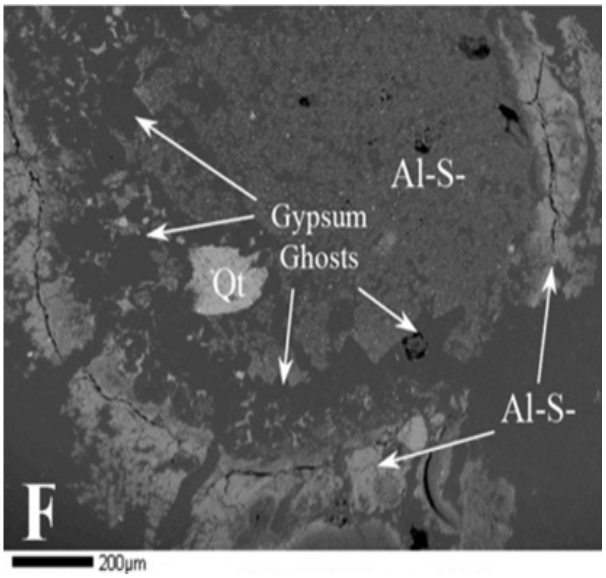


Figure 1-11. Limestone fragment containing gypsum ghosts.

Source: Thomas and Romanek (2002).

1.3.4 Relatively Fresh Substrate

Exhumation of bench and pilot scale BCR units in test programs typically reveals a deep zone of "relatively fresh" organic substrate deeper in the sulfide zone. This layer often exhibits the appearance and fabric of recently placed substrate despite the likelihood that some organic degradation has occurred. Gusek (personal communication 2011) reported observing green hay in the relatively fresh zone of the BCR at the Ferris-Haggerty mine site in Wyoming that exhibited the appearance similar to the substrate placed on the day the BCR was constructed (see Appendix B.4). This layer often exhibits the appearance and fabric of recently-placed substrate despite the likelihood that some organic degradation has occurred. During operation of a BCR, by the time the water reaches this zone, virtually all the metals have been precipitated into oxides, hydroxy-sulfates, sulfides, or carbonates. If the precipitates are formed in the pore space of the sulfide zone, ultrafine, colloidal-sized precipitates may attach to the surfaces of the organic materials and be retained. The extent of this phenomenon may vary from BCR to BCR and may increase with BCR maturity.

Gusek (personal communication 2011) measured metal concentrations in substrate exhumed from two locations in a pilot BCR at the Haile Gold Mine (unpublished data) in 2004 (see Appendix B.3). The unit had operated for about two years (Gusek and Schneider 2010). Substrate samples from the lower portions of this unit typically did not contain elevated levels of the parameters of interest (such as Al, As, Cd, Cu, Mn, and Zn). In fact, the concentrations of these parameters were typically less than the concentrations in a baseline, fresh archival sample of substrate. Only iron levels appeared to be elevated in the relatively fresh substrate layer compared to the baseline in one of the two sample locations (baseline = 7,380 mg/kg Fe; zone 18 to 24 inches = 10,100 mg/kg Fe).

2.0 DETERMINING THE APPLICABILITY OF A BIOCHEMICAL REACTOR

As discussed in Section 1.0, because no single treatment unit type works in every situation or with every type of MIW chemistry, a BCR is normally a single component within an overall sequential MIW treatment system (Gusek 2002; Gusek and Wildeman 2002). This section discusses the parameters that apply to and sometimes limit BCRs and also describes the general functions of pre- and post-BCR treatment components.

The applicability and construction of a BCR for MIW treatment depends on a number of site-specific factors. These factors are discussed here briefly and in more detail in Section 4.0 and Section 5.0. These factors include but are not limited to:

- net acidity, alkalinity, and pH of the MIW
- oxidation state (ORP/Eh) of the MIW
- metals and metalloids in the MIW
- flow of MIW (including seasonal and storm fluctuations)
- physical area (areally and topographically) available for treatment footprint
- access agreements and limitations
- climate and weather effects
- regulatory requirements, limitations, and land use restrictions
- cost considerations and limitations
- public acceptance, including odor and safety issues
- wildlife concerns and issues
- other pertinent site-specific issues

BCRs have successfully treated a wide variety of flows (from 1 to 1,200 gpm) and water chemistry (pH from 2.0 to 8.0; total metals from 0.4 mg/L to over 4,000 mg/L). Three sites in particular represent a variety of mining situations and climatic conditions:

- Ferris-Haggerty Mine (Appendix B.4) is an abandoned underground copper mine site in Wyoming at 9,500 feet elevation that has limited access due to cold weather.
- West Fork Mine (Appendix B.7) is an active underground lead mine in Missouri designed to treat 1,200 gpm to aquatic water quality standards as required by the National Pollution Discharge Elimination System (NPDES) permit. The system was built on 1996 and periodic maintenance is required to maintain treatment.
- Brewer Gold Mine (Appendix B.1) is a site in South Carolina that used the passive treatment of very acidic (pH = 3.1) heap leach solution at a gold mine. The mine has since been closed.

A BCR decision tree is presented in Figure 2-1. This decision tree presents the basic factors used to evaluate the efficacy and applicability of BCR treatment for a particular MIW. Each factor is discussed in the following sections. Each box of the decision flow process can be used to navigate,

and return, to the specific section of the Chapter and section discussing the importance of each topic.

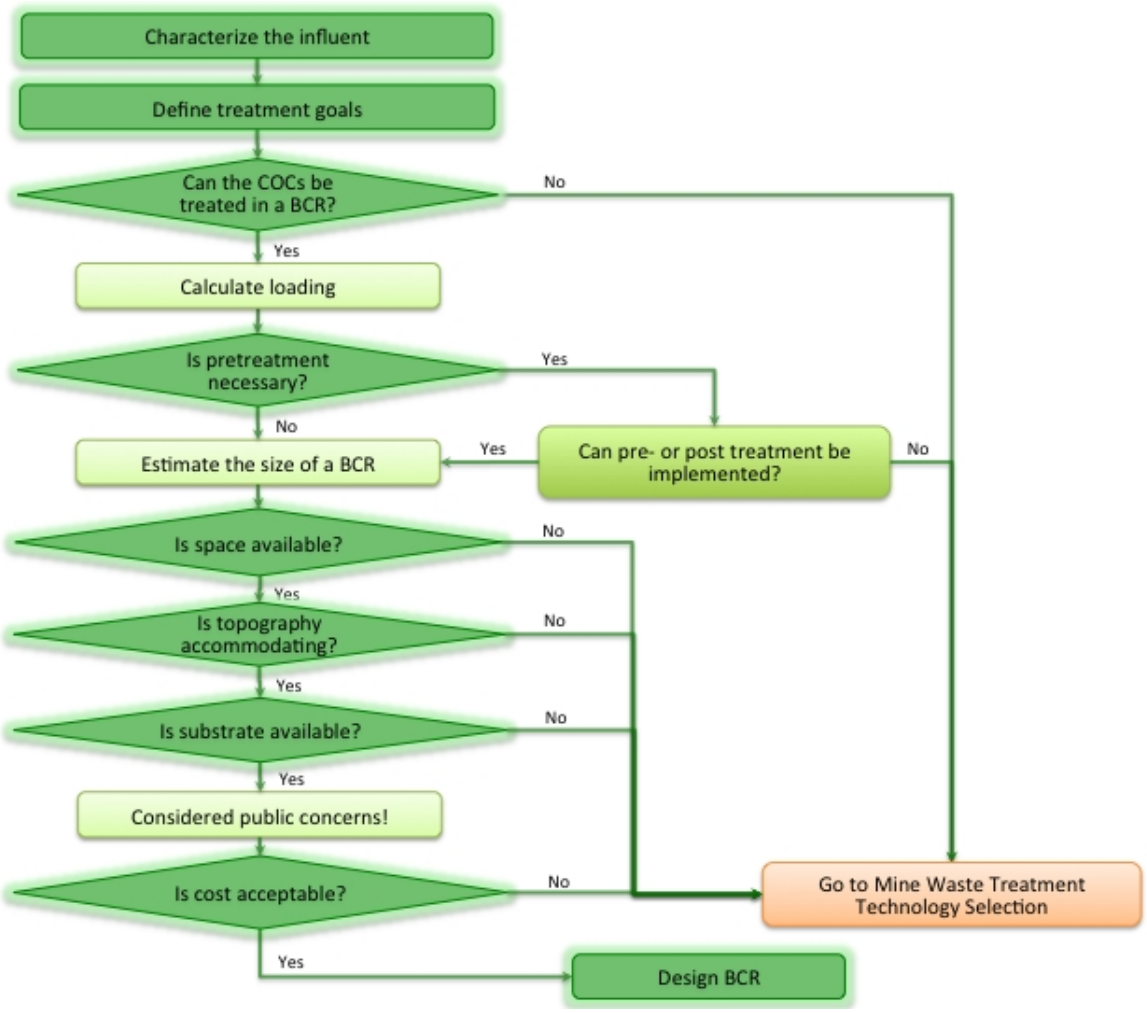


Figure 2-1. Decision flow process for determining the applicability of a biochemical reactor.

2.1 Characterize Influent

The first component of the decision tree involves assessment of the MIW characteristics. Because of the variability of MIW from site to site, particular site MIW must be accurately characterized in order to select and design an appropriate treatment technology. This characterization includes analysis of water chemistry parameters and flow rates and then calculation of contaminant loading (Section 2.4). Contaminant loading is used to determine pre- and posttreatment requirements and BCR sizing and design and are discussed further in Section 2.6 and Section 4.0. A comprehensive initial water chemistry analysis includes known and suspected contaminants of concern (generally metals and metalloids), major cations and anions, sulfate, alkalinity, acidity, total dissolved and suspended solids, redox conditions, and pH (see Table 6-1 for further information on sampling methods).

As the influent is characterized, consider seasonal fluctuations in chemistry and flow in response to climate. Characterizing the MIW may require seasonal monitoring of flow and chemistry for a year or more. Peak flow and loading conditions should be well understood, since these are factors that affect the size and required components of the BCR system. Since peak flows and loading may change year-to-year, gather as much information on annual changes as is practical. Evaluation of historical regional stream flow data and climate data (such as that collected and published by United States Geological Survey (USGS), National Oceanic and Atmospheric Administration (NOAA) or others) may be useful in assessing overall trends and predicting peak flows and seasonal flow fluctuations if actual site data is not available.

2.2 Define Treatment Goals

The treatment goals and discharge requirements must be well defined to properly design a BCR within a sequential treatment system. Generally the overall treatment goals of the system are defined by the regulatory agency responsible for overseeing or permitting the site where treatment is planned. The treatment goals specific to the BCR unit are determined in conjunction with selecting and designing pre- and posttreatment technologies that will form the sequential treatment system.

The overall goals for a specific treatment system depend on the geographical location and receiving water body, as well as whether the site is an abandoned or active mine site. Generally it is assumed that a treatment system discharges effluent to surface water rather than groundwater. Surface water discharges to streams, wetlands, and oceans are typically regulated by the Clean Water Act of 1972 (CWA) or state or tribe equivalent. Discharges to groundwater may be considered as an alternative and may have different regulatory requirements. In most cases, active mine sites must comply with provisions of Surface Mining Control and Reclamation Act (SMCRA) for coal mining and the National Pollutant Discharge Elimination System (NPDES); see the USEPA NPDES Permit Program Basics web page (http://cfpub.epa.gov/npdes/home.cfm?program_id=45) for more information. For hard rock mining, sites must secure permits for discharges to surface waters. Allowable contaminant concentrations under NPDES permits are determined by USEPA's technology-based standards, maximum contaminant levels (MCLs) or toxicity-based aquatic life standards established for stream uses, or the state or tribal equivalent. The permits are generally site specific and may include total maximum daily loads (TMDL) to determine the treatment effluent goals. A specific point of compliance (POC) is normally established during the permit process (Section 6.2.3).

Regulatory flexibility depends on the purpose of a treatment system, including whether the system is intended as an interim water treatment measure or a long-term treatment solution. Regulatory flexibility also depends on whether the system is intended to provide at least some level of treatment in an abandoned, remote location that is otherwise impractical for more active water treatment systems (see ITRC MW-1 2010 Regulatory Issues , http://www.itrcweb.org/miningwaste-guidance/reg_issues.htm and Section 6.0 of this document).

Often when BCR systems are used as an interim water treatment measure, (while a long-term plan is being formulated) certain water quality parameters may be waived that would otherwise be

applied to treatment plant discharges. Often only pH and metals are targeted for regulatory compliance in these interim or very remote situations (see Appendix B.8, Leviathan Mine Case Study and Appendix B.2, Mayer Ranch Case Study). Depending on the setting and goals of the system it may be possible to target percent contaminant removal for the system. However, when a BCR system is planned as the long-term solution in an accessible area, regulators may target water quality parameters that are more comprehensive and may require it to meet tribal, state, or local water quality criteria or MCLs.

Regulatory treatment goals normally apply to the final effluent of the sequential treatment system, which includes the BCR. The treatment system may contain pretreatment such as a settling pond to reduce the sediment load (Section 2.4) prior to a BCR, or posttreatment unit such as an aerobic wetland (Section 2.5). However, each of the components of the treatment system, including the BCR, must have an established treatment goal in order to accurately design the BCR and required pre- and posttreatment units for successful installation and performance. Establishing treatment goals for individual process units for the treatment system is done in the design phase. Well-defined expectations also enable the operator to develop an appropriate intersystem monitoring program to identify and resolve potential problems before damage to the system occurs.

2.3 Evaluate Treatment Effectiveness for Contaminants of Concern

A particularly useful tool for evaluating which elemental contaminants can be removed in passive treatment, and BCRs in particular, is the Periodic Table of Passive Treatment for MIW developed by Gusek (2009) and shown in Figure 2-2. Elements shown in blue are removed in anaerobic conditions, which are produced in BCRs. Orange indicates elements that are removed in aerobic conditions that may be designed in pre- or posttreatment units. Red indicates elements that are not removed in passive treatment. These elements in red contribute to total dissolved solids (TDS) in the treatment system effluent.

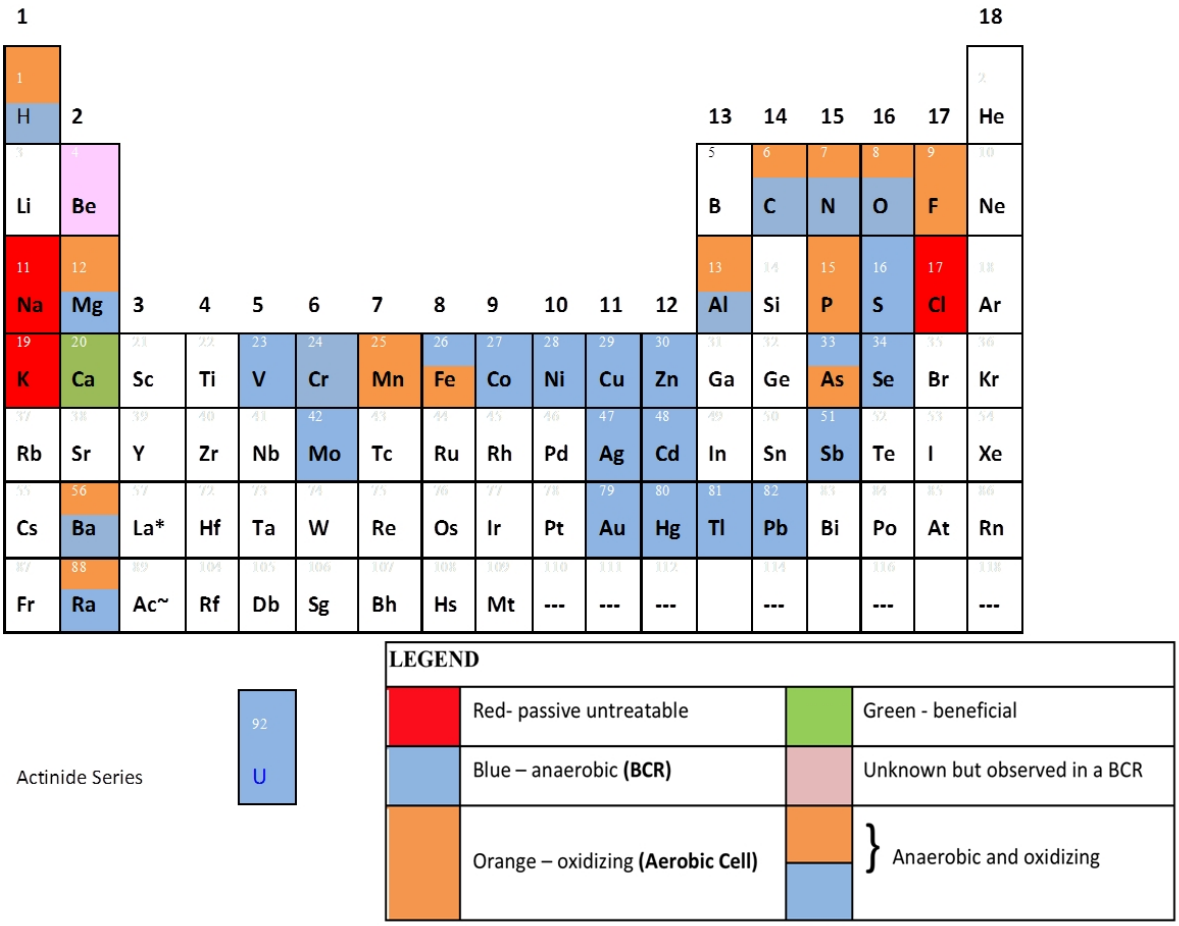


Figure 2-2. A periodic table of passive treatment for MIW.

Source: Reprinted with permission and additional modifications from author (Gusek 2009).

TDS are not removed in BCRs and in some cases certain TDS components may increase due to alkalinity addition or other amendments in the BCR or pre- and posttreatment units. While calcium is also conserved in BCRs and may actually increase if limestone is a reactor substrate component, calcium is depicted in green because its beneficial contribution to hardness diminishes the toxicity of some metals to aquatic life.

BCRs generally are designed to reduce sulfate to sulfide to promote the formation of metal sulfide precipitates, which are removed as amorphous solids from the MIW. Sulfate in the influent water is a necessary component for metals removal by BCR treatment. The minimum concentration of sulfate required to remove the desired concentration of metals can be calculated stoichiometrically from the sulfide reduction and the metal sulfide precipitation equations (Table 1-2 and Gusek 2002). A contingency is then added to account for sulfate or sulfide lost to gaseous hydrogen sulfide formation, adsorption, and inefficient transformation. In most MIW waters, sulfate is stoichiometrically in excess of metals and is not removed by a significant percentage. Certain sites may

only see a 10 to 20% decrease in sulfate concentration in the effluent (see Appendix B.8, Leviathan Mine Case Study). However, in certain MIW waters, low sulfate concentrations may limit the metals removal capacity and preclude treatment by a BCR. Additionally, biological sulfate reduction to sulfide can be the rate-limiting step in the process if conditions are not favorable for this biological activity.

As shown in Figure 2-2, a BCR can remove both aluminum and iron by aerobic and anaerobic processes. Often these metals are in very high concentrations in proportion to other metals in the MIW. Therefore, calculate the loading of these metals because they can contribute to rapid clogging of the BCR, require a larger size BCR, require pretreatment, and shorten the life span of the system. When high concentrations of iron or aluminum are present in the MIW, consider pretreatment steps to reduce this load to the BCR. Pretreatment to reduce metal concentrations can allow smaller sizing and longer design life for the BCR. Various pretreatment technologies are described further in Section 2.5.

BCRs can also treat other compounds in MIW. Cyanide, from gold leaching processes, is degraded biologically in the anaerobic conditions of a BCR. Nitrate and nitrite residuals from blasting are biologically reduced to nitrogen gas in BCRs. Ammonia, also a residual from blasting, may be taken up by the bacteria or plants if they are present in a BCR, but is most effectively removed by oxidation to nitrate in aerobic conditions prior to or within a BCR. Also, total suspended solids (TSS) may be removed to varying degrees by interception within the BCR, although TSS removal may be greater in BCR with solid phase substrates (see Section 3.0).

Effluent from BCRs typically contains residual soluble organic substrate and reduced compounds (sulfur forms), and dissolved oxygen (DO), that can adversely affect water quality. Biochemical oxygen demand (BOD) and chemical oxygen demand (COD) can substantially increase from BCR treatment because of the dissolution and transport of the organic material added as microbial substrate. Additionally Butler et al. (2011) demonstrated toxic BCR effluent by aquatic toxicity testing from four different treatment systems. The effect of discharging effluent with these characteristics should be weighed against the benefit of treatment for the other constituents and the possible ecological receptors in the receiving water body, which could be impacted. Posttreatment steps should be considered to remove these compounds and raise dissolved oxygen levels prior to final discharge for most sites.

Hydrogen sulfide may be generated in a BCR and can present occupational health and nuisance concerns. Hydrogen sulfide can accumulate to toxic levels in certain circumstances such as under a BCR cover. The Occupational Safety and Health Administration (OSHA) has set an acceptable ceiling limit at 20 parts per million (ppm); other organizations recommend varying levels. Concentrations in air above 500 ppm may result in death. At very low concentrations no health effects have been found in humans, so at low concentrations the gas is considered only a nuisance due to the rotten egg odor. See the hydrogen sulfide fact sheet from the Agency of Toxic Substances and Disease Registry (ATSDR) for more information (http://www.atsdr.cdc.gov/tfacts114.pdf). The design process must address the generation of hydrogen sulfide gas since proper design and siting of the BCR can mitigate this problem. Methods of H_2S control include aeration, blending untreated

water with effluent, chemical oxidation, and pH control (neutral). Hydrogen sulfide gas can also create a corrosive environment in the immediate proximity to the BCR and corrode metal components of equipment.

2.4 Calculate Loading

Calculating the loading to the treatment system is a key step in determining what type of pre- and posttreatment is required. Additionally, loading information is used in designing and sizing the BCR to estimate the expected operating life of the system. These design aspects are discussed further in Section 4.0.

2.5 Determine Pre- and Posttreatment Requirements

Design of a BCR is presented in detail in Section 4.0, and pre- and posttreatment system design is addressed elsewhere (INAP 2010; ITRC MW-1 2010). This section presents limitations to a BCR and describes how pre- and posttreatment of MIW can be used in conjunction with a BCR in light of those limitations. Assessment of influent water quality and flow is necessary in order to evaluate if a BCR can treat the MIW directly, and if not, whether pretreatment can be implemented. Pretreatment may be necessary for the influent MIW (prior to BCR treatment) to neutralize acidity and pH, reduce concentrations of particular metals (such as iron and aluminum), remove suspended solids, and change the water oxidation state. Pretreatment approaches may be passive, semi-passive, or active treatments, depending on site conditions. Active processes require energy from outside of the system, while no energy is added for passive processes. Semi-passive processes can be some combination of active and passive, or be powered by renewable energy (such as wind, solar, or hydroelectric). Examples of primary pretreatment processes are as follows:

- oxidation
- neutralization
- sedimentation

If used, pretreatment processes are part of the overall influent MIW management system. The decision tree shown in Figure 2-3 illustrates pretreatment processes. As shown in this figure, acidic MIW with high aluminum and iron concentration might be treated without using a BCR due to clogging issues. In addition to potential clogging, MIW BCRs are sensitive to widely variable influent flow and chemistry changes. A water storage facility can be used to manage the influent MIW volume and provide a consistent flow to the BCR unit or other pretreatment system. Although a storage pond in itself is not a pretreatment, it may be an important component of the sequential treatment system. The pretreatment approaches described below can be used prior to or after the water storage facility.

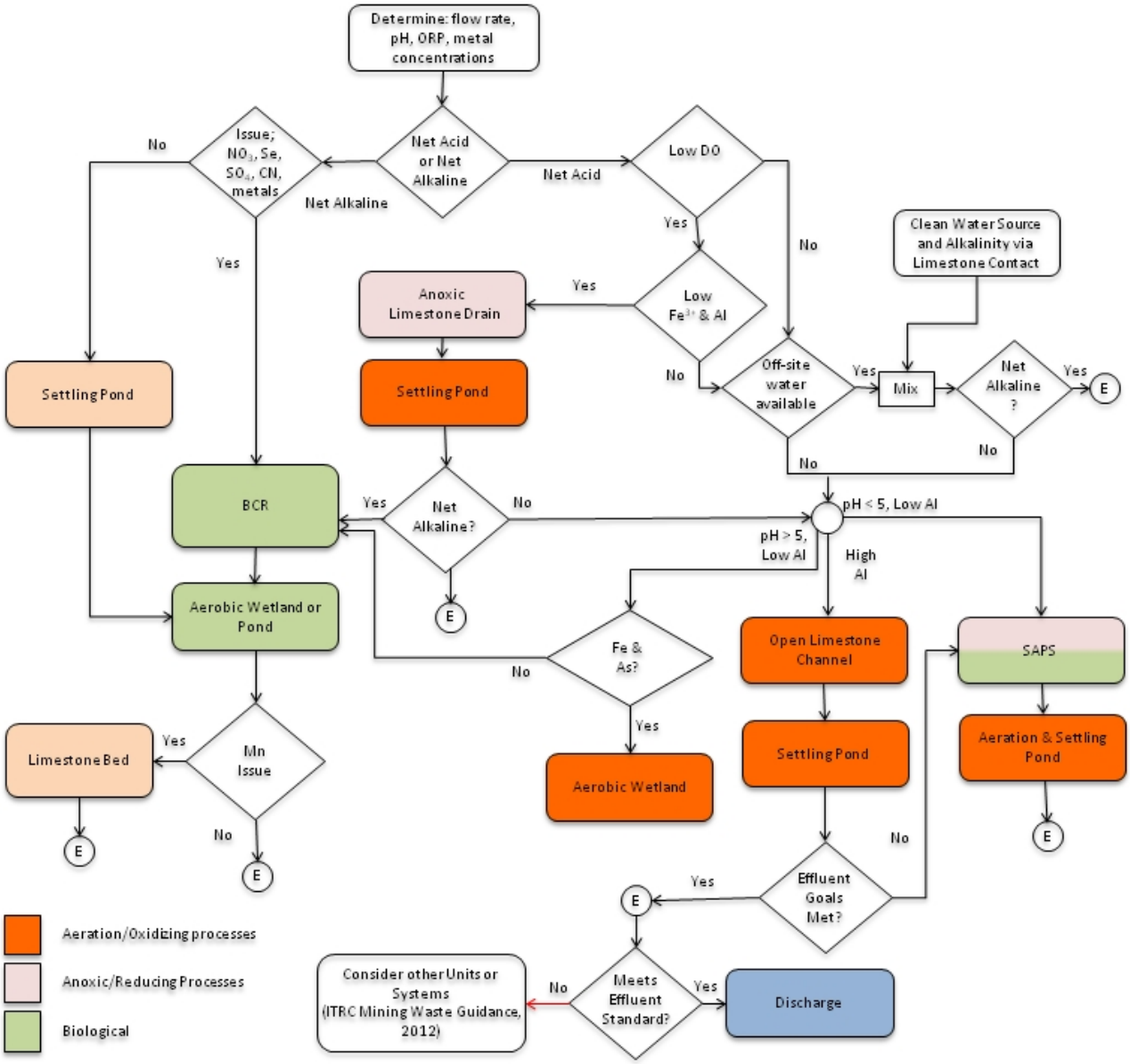


Figure 2-3. Decision tree for pre- and posttreatment units for BCR systems.

Source: Adapted from Gusek (2008).

Posttreatment of water from the BCR unit typically polishes the water prior to final discharge. Although metals have been removed, treated MIW that leaves a BCR unit can have low DO, high concentrations of suspended solids, BOD/COD, and reduced sulfur forms. The most common posttreatment methods for BCRs involve combinations of oxidation, sedimentation, pH adjustment, and aerobic wetlands.

2.5.1 Oxidation

The purpose of oxidation is to increase the oxidation state of the flow entering or leaving a BCR. For influent MIW this process converts various metals that may be present in a reduced form to a more oxidized form (such as Fe^{3+}), allowing these metals to precipitate as oxides or oxy-hydroxides. Oxidation can also help release excess CO_2 from the water to the atmosphere, which can result in a slight increase in pH (Younger et al. 2002; Stumm and Morgan 1996, INAP 2009). For treated effluent from a BCR, oxidation increases DO and reduces concentrations of sulfides, BOD, and COD. Oxidation can be conducted by physical processes that add oxygen to the system from air (aeration), or by chemical oxidation processes (addition of chemical amendments), or by aerobic wetland systems. Oxidation of MIW often produces solids, which may require sedimentation in a settling pond prior to or after the BCR (see Section 2.5.3).

2.5.1.1 Aeration

Aeration processes can be active or passive. Active aeration can be completed within an MIW storage pond, tank, or pipe using a mechanical mixing device or by injection of air or oxygen. Passive aeration can be completed by routing MIW over cascades, riffles, or along roughened channels. This type of aeration requires little energy, but is only applicable to sites that have the appropriate topographical conditions. The energy can often be supplied with wind or solar energy generation on site (Appendix B, Mayer Ranch Case Study). INAP's GARD Guide (2010) provides examples of numerous oxidation processes for MIW, as does PIRAMID (2003). Aeration can also be completed using trickling filters or by chemical oxidation methods. Trickling filters operate by feeding water through the top of a tank packed with high surface area media (Younger et al. 2002). An active commercial treatment system that aerates water with the use of a rotating drum is the Rotating Cylinder Treatment System™ (RCTS™) described in ITRC MW-1 (2010).

Because a BCR operates in a reduced oxidation state, oxidation of MIW prior to BCR treatment may not be desired. However, oxidation of MIW may be a component of other necessary pretreatment approaches such as alkaline addition, sedimentation, or aerobic wetland treatment. In some cases, the MIW may already be in an oxidized form due to site-specific conditions. When the oxidized MIW has high iron concentrations at or near saturation, further aeration can convert more of the iron to the oxidized form and maximize the formation of ferric iron precipitates. BCR effluent can be oxidized using methods similar to those mentioned above, but oxidation is most commonly achieved with constructed aerobic wetlands and settling ponds.

2.5.1.2 Constructed Wetlands

Constructed aerobic wetlands typically consist of shallow excavations filled with flooded gravel,

soil, and organic matter to support wetland plants such as *Typha*, *Juncus*, and *Scirpus* sp. (Skousen et al. 1998). Aerobic wetlands promote oxidation and hydrolysis in the shallow surface water, which causes precipitation of metal oxyhydroxides and carbonates that form under aerobic conditions. This process results in uptake of metals by plants and physical filtration of precipitates by plant root matter and biomass. Unlike typical BCRs, aerobic wetlands are most often horizontal flow systems. Extensive published literature on aerobic wetlands documents decades of successful treatment of certain types of coal-mine drainage. ITRC WTLD-1 (2003) provides a detailed analysis of the application, contaminant removal mechanisms, and regulatory guidance of constructed wetland systems, while ITRC MW-1 (2010) provides additional guidance on the technology as applicable to MIW treatment. The term "constructed wetland" as described by ITRC MW-1 (2010) and others (such as Hedin et al. 1994 and Skousen et al. 1998) includes aerobic and anaerobic processes, which are processes that are now defined as BCRs according to this guidance. This guidance avoids the use of the term "wetland" to describe a BCR system because of regulatory issues that can be associated with this term.

Two applications of aerobic wetlands include pretreatment of MIW prior to a BCR unit or posttreatment of MIW after a BCR unit. As a pretreatment, aerobic wetlands are typically used to treat net-alkaline water to promote removal of iron, aluminum, and manganese as oxy-hydroxide precipitates. The system is designed to form these precipitates and sustain plant growth that provides retention of the precipitates.

As a posttreatment to BCR units, aerobic wetlands can be used to polish the BCR effluent. The aerobic wetland provides an increase in DO, uptake of BOD, and retention of solids from the BCR effluent. Plant uptake can also help to remove trace metals in the BCR effluent. This posttreatment step is also called an aerobic polishing cell (APC). In conjunction with proper sedimentation processes, the aerobic polishing unit can produce a final effluent water quality that is acceptable for discharge to receiving streams or other water bodies.

2.5.2 Neutralization

Neutralization uses an alkaline reagent to raise the pH of MIW and remove metals via precipitation of various metal oxides, hydroxides, and hydroxysulfate forms. Alkali materials include hydrated lime ($Ca(OH)_2$), quick lime (CaO), sodium hydroxide (NaOH), sodium carbonate (Na_2CO_3), magnesium oxide (MgO), limestone ($CaCO_3$ or dolomite, and waste products such as fly ash, lime kiln dust, and slag. Several references describe in detail alkaline addition approaches, such as Casey (1997), Younger et al. (2002), INAP (2010), Skousen et al. (1990), among many other texts. As a pretreatment to a BCR, alkaline addition can reduce the acidity of MIW, which results in a reduced BCR unit size required to treat the MIW. After alkaline addition, a BCR system follows to further neutralize the MIW and remove trace amounts of metals. Alkaline addition processes typically require a combination of oxidation, alkaline addition, and sedimentation (Younger et al. 2002).

2.5.2.1 Active and Semi-Passive Neutralization

Conventional active alkaline treatment for MIW is typically implemented by addition of reagents

such as hydrated lime, limestone, or sodium hydroxide. This process is considered active because both the chemical processing and mixing require controlled addition of mechanical energy. The reaction rate at which metal precipitates form depends on numerous factors such as the MIW chemistry, type of alkaline reagent and dosing rate, reaction temperature, and degree of mixing and oxidation. Oxidation is implemented throughout the alkaline processes to improve the precipitation rate. Sedimentation is implemented after the precipitation and mixing process to settle the precipitates from solution.

An important component of application of alkaline addition is the solubility of various metal precipitate forms. The solubility is specific to each metal and controls the stability of the metal either in the oxidized solid precipitate form or in the dissolved form. Because many MIWs contain trace metals that are highly soluble at neutral pH (such as cadmium, lead, nickel, manganese, and zinc), conventional alkaline addition may require a pH increase to 9.0 or greater in order to ensure that the targeted metal removal efficiency is achieved.

Semi-passive alkaline addition can be implemented by passing the MIW through a solid reagent (such as sodium hydroxide pellets or lime-based solids) or addition of a solid or liquid reagent to a MIW stream or pond. Semi-passive approaches have also been implemented that use micro-hydropower (water wheels) to feed solid and liquid neutralization reagents at set rates into acidic MIW. Two commercial systems that feed alkaline reagents at a rate defined by the water flow are Aquafix™ solid pebble lime dosing and Wheeltreater™ sodium hydroxide dosing (Gusek and Conroy 2007).

Semi-passive alkaline addition to a high endpoint pH (9.5) to remove metals can be difficult to implement due to factors that affect the efficiency of the process, as well as the difficulty of managing the very large volume of sludge produced. Alternatively, semi-passive approaches can raise pH of acidic MIW to a moderately acidic pH at which iron and aluminum precipitate (such as a pH of 4.5). This approach removes a majority of the iron and aluminum that otherwise might cause plugging in a BCR unit. However, regular operation and maintenance must be performed to manage the metal precipitate sludge produced.

2.5.2.2 Passive Neutralization

Passive alkaline addition is generally implemented by allowing the MIW to flow through a bed of limestone or dolomite; at most sites considered for passive treatment systems the low cost and ready availability of limestone or dolomite are unmatched. Some low-cost waste products can also provide alkalinity, such as fly ash, lime kiln dust, and slag. Waste products should be evaluated for unanticipated contaminants prior to use.

Examples of passive treatment units are the anoxic limestone drain (ALD, http://www.itrcweb.org/miningwaste-guidance/to_anoxic.htm), open limestone channel, limestone bed, and a reducing alkalinity producing system (RAPS). As acidic water contacts limestone, the limestone dissolves, releasing calcium and bicarbonate, which raises the pH. Because the reaction works by dissolution, the limestone eventually becomes depleted and requires replacement. However, a coating of

limestone by ferric iron, aluminum, and manganese precipitates can occur in oxidized MIW very quickly, rendering the limestone ineffective.

MIW can be highly oxidized, so a RAPS system prior to a BCR unit may be appropriate. This type of system reduces the oxidation state of the water using an organic layer and provides some neutralization using a limestone layer. A RAPS can also be used after an oxidation or neutralization pretreatment process, both of which create a highly oxidized MIW. Approaches for design and examples of RAPS are provided in Younger et al. (2002), INAP (2009) and Gusek and Figueroa (2009), Kepler and McCleary (1994), and Watzlaf et al. (2000).

A downflow RAPS unit, also called a vertical flow pond (VFP), consists of a layer of limestone overlain by a layer of compost or other organic substrate material. The purpose of the organic layer is to strip dissolved oxygen from the water and to reduce any Fe^{3+} to Fe^{2+}prior to the water contacting the limestone aggregate (Younger et al. 2002). If a RAPS system were designed to be upflow, then the layers of organic material and limestone would be reversed. A SAPS uses the same principles, except the term "successive" indicates that several alternating layers of organic material and limestone are used to treat the MIW (Younger et al. 2002) or that more than one RAPS is used in series.

2.5.3 Sedimentation

Sedimentation processes remove solids from suspension. As pretreatment to a BCR, sedimentation removes or reduces the suspended solids in the influent MIW. The sedimentation step may be part of the influent MIW management system, such as a water storage pond that also achieves sedimentation. As described above for oxidation processes, some MIW types may also contain high ferric iron concentrations, which result in formation of ferric iron precipitates. Neutralization processes also form a high volume of sludge. For either oxidation or neutralization processes, sedimentation is required to remove these suspended solids prior to BCR treatment or any other treatment process.

Sedimentation ponds or conventional clarification processes must provide adequate residence time, and therefore low enough overflow velocity, to allow settling of suspended solids. Basic design sizing for a sedimentation pond to remove inert suspended solids is provided by Younger et al. (2002). Sedimentation ponds can be operated passively, however, sufficient space for operation of the system must also include sludge storage volume.

An important component of sedimentation processes is collection and management of the settled solids or sludge. Design sedimentation systems to provide for ongoing maintenance and removal of settled solids as needed. Treatment sludge may be a hazardous waste under RCRA (40 CFR Part 261.24 Table 1) and should be tested using the toxicity characteristic leaching procedure (TCLP) and any applicable tribal or state required tests to determine the disposal requirements. Recycling and reuse options may be available for the sludges (ITRC MW-1 2010). On-site disposal of non-hazardous sludge may be possible, depending on site-specific conditions (such as dewatering, space requirements, and costs).

Sedimentation of MIW as a posttreatment addition to a BCR unit reduces the suspended solids content of the effluent. As part of the overall passive treatment system, sedimentation of BCR unit effluent can be implemented using an open pond or aerobic polishing unit. The pond provides the retention time to settle suspended metal sulfides or other sulfur precipitate forms (for example, elemental sulfur) that form within the BCR and are present at some quantity in the effluent stream. The sedimentation pond also provides oxidation of the effluent MIW at the surface. Deeper ponds can be used to provide adequate retention time and as a control to limit re-oxidation of reduced sulfur forms. A sedimentation pond may also be used as pretreatment.

2.6 Evaluate Site Environment and BCR Size

One of the key considerations when choosing a BCR is available space. Based on team experience, solid-phase substrate BCRs may require more space than liquid-phase substrate BCRs to achieve a higher hydraulic residence time with a comparable bed depth. Flow and water quality must be adequately characterized in order to estimate the size of the BCR. Free software from OSM AMD Treat (2008) at http://amd.osmre.gov can provide an initial estimate of BCR size. Additional details on reactor size and design are described in Section 4.0. When evaluating a site, additional space may also be needed for pre- and posttreatment processes (see Section 2.5).

Gravity flow BCRs are preferred, although for small flows solar powered DC pumps may be acceptable. In order to avoid pumping water, the site topography must accommodate the design. If the water source is an underground mine, a bulkhead might be used to partially flood the workings to provide enough static pressure head to deliver the water to a treatment site topographically above the mine portal. If space is limited immediately adjacent to the mine portal, consider installing a gravity pipeline to a flatter and more accessible area for a solid-phase substrate BCR. Alternatively a liquid substrate BCR may be used to overcome space constraints. The design of a liquid substrate BCR must address generated solids retention.

Initially BCRs were not used in areas with severe winter climates. However, by including covers in the design, BCRs have been used at high elevations and in areas with long and cold winters such as Colorado (Reisman et al. 2008), and Montana (Reisman et al. 2003). The covering should be vented, as it can cause gas build-up underneath. Additionally, a cover can keep rainwater and snowmelt from introducing higher electron acceptors and from diluting the BCR influent. Covering also can minimize evaporation in hot and dry climates (Buchanan and Gusek 2011).

2.7 Evaluate Substrate

A wide variety of organic materials have been used in BCRs (see Section 3.1). Most BCRs are constructed using solid-phase substrates, although liquid-phase substrates may also be used (ITRC MW-1 2010, Nacimiento Mine Case Study). Spent mushroom compost (SMC) was one of the first solid-phase substrate materials and continues to be widely used in the eastern U.S., where it is readily available (Prieto et al. 2008 and Venot et al. 2008); see Section 3.1.1. Newer systems, however, have mixed the SMC with wood chips in an attempt to increase the organic carbon content and permeability, and to extend system life (Eppley and Gusek 2010). Corn stover (leaves and stalks) was

shown in tests to be a good organic carbon source and is readily available in the Midwest (Ruiz et al. 2008). Any local agricultural or extension agency publication that discusses the relative compostability of organic materials is useful in selecting appropriate locally available solid substrate materials. The lower the C:N (carbon/nitrogen) ratio, the more susceptible the material should be to organic decay; higher C:N ratio materials like wood (200 to 1,100) should provide a slow release source of carbon.

2.8 Evaluate Cost

If water chemistry, site, and likelihood of acceptance by the owner and the public indicate a BCR may be applicable at a site, then next evaluate the cost of the BCR. AMDTreat (OSM 2008) provides default direct cost estimates for BCR materials and overall construction. While engineering costs can be input as a percentage of construction in one AMDTreat module, other indirect costs such as construction management and contingency must be determined and input as separate cost items.

The money spent on the evaluation, testing, design, operation, monitoring, and maintenance of a BCR, or a larger passive treatment system for MIW consists of two elements: material cost and service cost.

Materials costs such as land, organic substrate, limestone, piping, valves, drains, monitoring instrumentation and materials, and many others are driven by availability, placement and replacement, and are easily predicted in the short term. These costs have unit prices (such as dollars/yd^3 for substrate, dollars/yd^2 for liner material, and dollars/unit for sampling instrumentation). It is relatively easy to estimate the quantity of each that is necessary for the BCR installation and maintenance, plus a reserve for expected materials and maintenance.

The service costs, on the other hand, are difficult to predict, value, and control. To travel to, retrieve, transport, and place substrate is an additional cost per cubic yard. Implementation according to the design specifications and quality assurance checks along the way also add to costs. Compare this cost to using improper substrate that may sit in storage for a year waiting for construction to catch up with material availability, be incorrectly placed and replaced, treatment delays, inactive downtime and the difficulty in accurately pricing becomes readily apparent. Successful installation and operation requires proper material, testing, skilled services, careful construction, and experienced operators.

Table 2-1 is a checklist of elements that should be considered when developing a cost estimate for a BCR. This table is a guideline checklist, but is not comprehensive for all sites. Descriptions or rows can be added or removed after careful consideration. Competitive cost and experienced-based prices should be obtained when available.

Table 2-1. Example of a worksheet for cost estimates for a biochemical reactor

Cost Item	Description	Estimated Material Cost	Estimated Service Cost	Estimated Total Cost
1	Site Evaluation			
	Loading			
	Site construction characteristics			
	Material availability			
2	Testing			
3	Design			
4	Mobilization and Demobilization			
5	Diversion and Care of Water During Construction			
6	Dewater Impoundments			
7	Excavation			
8	Grading			
9	Substrate Materials (multiple options)			
10	Calcareous Rock Lining			
11	Liner Material			
12	Water Control Systems			
	Piping			
	Valves			
	Drains			
13	Monitoring			
	Instruments			
	Installation			
14	Construction (treatment material and piping)			
15	Erosion Control			
16	Roadway Grading and Material			
17	Fencing			
18	Seeding			
	Bed preparation			
	Agricultural limestone			
	Fertilizer			
	Seed			
	Mulching			
	Planting			
19	Administrative Facilities			
20	Operation			
	Start-up			
	Routine operations			
21	Monitoring			
	Data collection			

Cost Item	Description	Estimated Material Cost	Estimated Service Cost	Estimated Total Cost
	Data storage			
	Reporting			
22	Maintenance (expected substrate longevity- X years)			
	Substrate replacement			
	Sludge management			

2.9 Public and Tribal Concerns and Acceptance

Although the BCR technology has significantly progressed through the years, public and tribal concerns with the technology remain (see Section 8.0). These concerns include several major issues (see B.1) and are discussed further in Section 7.0.

- Surrounding land use: A BCR sited in a remote area with no nearby residences generally has few public acceptance issues. In a populated area, even if in rural use, public concerns should be considered and mitigated.
- Hydrogen sulfide gas odor and toxicity: Hydrogen sulfide gas is produced in BCRs in varying levels. Ensure that the BCR does not expose surrounding residences or workplaces to toxic levels of hydrogen sulfide gas. Be aware that hydrogen sulfide gas emissions below the toxicity level can still result in nuisance odors (rotten egg smell) to surrounding properties.
- Aesthetic issues: Often the appearance of the BCR and pre- and posttreatment units concerns surrounding land users. Some BCRs are designed to look like wetlands areas and some are designed with plastic covers. Consider the appearance for public acceptance when siting BCRs in populated areas.
- Accessibility, trespassing, and safety: The accessibility of the site can influence the likelihood of trespassing and public safety. Consider the safety of open ponds, which may attract children or curious adults in residential areas. Open ponds that are lined with plastic liner can attract animals as well and may create trapping and drowning hazards for animals and humans. Modify the designs as needed to limit access and prevent trapping hazards for humans and animals.

With proper testing, design, construction, maintenance, and monitoring, BCRs can achieve public and tribal acceptance.

2.10 BCR Test Design

Once the determination (based on the decision tree flow diagram) is made that a BCR system can treat the MIW in question, the project can move into a phased design. Phased design includes testing of possible substrates, proof of principle testing, bench scale testing, and pilot testing to develop information for the detailed system design. The approaches for testing phases are described in detail in Section 3.0.

3.0 TESTING PLANS, DESIGN, AND PROTOCOL

The design of passive treatment systems is an inexact science due to the variety of site-specific and changing water chemistries requiring treatment and to the variety of materials that can be used in construction. For chemically simple coal drainage (relatively mild pH water containing iron and manganese and little or no aluminum), engineers and scientists at the former U.S. Bureau of Mines developed standardized or "cookbook" design criteria (Hedin, et al. 1994) for aerobic systems that are still being followed today. Wildeman et al. (1993) developed a phased design protocol that is appropriate for more complex acidic chemistries, as well as neutral to net alkaline drainage chemistries. These two approaches are opposite ends in a design philosophy continuum.

At one end of the design continuum, the danger in any cookbook design approach is that it may not properly address conditions other than those that were originally used to develop the standardized design criteria. For example, the treatment of low pH water containing dissolved aluminum is especially problematic and outside the original U.S. Bureau of Mines design criteria, which addresses the issue by restricting the application of anoxic limestone drains (ALDs). A precise and reliable aluminum design guideline has yet to be developed for ALDs and probably should not be considered due to the complexity of aluminum chemistry. While iron can be precipitated aerobically as ferric hydroxide or anaerobically as a sulfide or carbonate, the list of aluminum mineral species found in nature (and thereby possible in a passive treatment system) is extensive. If heavy metals are present (for instance, in MIW from metal mines), then the design challenges multiply rapidly. Cookbook approaches become futile for passive treatment design when the effect of varying anionic concentrations and water temperature are added to these complications.

At the other end of the design spectrum, mining, chemical, and other industries have long used a phased design process. The concept is simple: start small, learn from failures, and build on successes until the data required to properly design a full-scale treatment system are available. These data significantly reduce the risks of the full-scale system failure or less than optimum performance. Wildeman, et al. (1993) propose a design protocol that includes laboratory-, bench-, and pilot-scale phases. This approach has been used at dozens of MIW sites.

A phased-approach design project typically begins in the laboratory with static tests, then graduating to final testing phases (bench and pilot) performed at the site on the actual MIW. Bench-scale testing determines if the treatment technology is a viable solution for the MIW and narrows initial design variables for the field pilot test. A proper bench-scale test reduces the duration of the more costly field pilot test. Field pilot test duration can range from days to years, depending on the nature of the technology. Depending on the equipment and personnel needed, significant costs may be incurred during the field pilot tests (mostly for sampling and analysis); however, these costs may be significantly less than active treatment pilot tests. This section describes a phased approach, with suggested test phases:

1. Identify and select solid or liquid substrate candidates (see Section 3.1).
2. Measure bulk substrate properties (see Section 3.2).

3. Conduct bench-scale quick paste tests (see Section 3.3.1).
4. Conduct bench-scale four-week bottle testing (see Section 3.3.2).
5. Conduct small-scale (barrel-scale) three-month or longer test with flow (see Section 3.4).
6. Conduct pilot-scale field test (see Section 3.5).

Detailed descriptions of testing phase activities are provided in the following sections.

3.1 Selecting BCR Substrate

As discussed in Section 2.8, local availability of solid or liquid substrate material sources often lowers the cost of the substrate mixture. In an ideal situation where a wide variety of candidate materials are available, selections of individual components (and their relative abundance in the mixture) can be influenced by their roles in the overall process. A BCR substrate is typically a blend of materials that together perform multiple roles as summarized in Table 3-1. BCR substrate may be a combination of solid and liquid substances as described in Section 2.7.

Table 3-1. BCR substrate functions and common components

Role	Examples of Components*
Source of long-term electron donor (food)	Wood chips, sawdust, walnut shells, mushroom compost, other composts, chitin, rice hulls
Source of short-term electron donor for start-up (fast food)	Manure, hay, straw, chitin, yard waste, brewery waste, beet pulp, corncobs, dairy whey, acetic acid, ethanol
Source of microbial inoculum	Ruminant manure, other BCRs, publicly owned treatment works (POTW) sludge, sludge pond or slime dams at mining sites, or septic system products such as RidX®, manure from browsing animal has a higher ability to break down cellulose
Source of alkalinity	Limestone, seashells, fly ash, cement kiln dust
Bulk to allow for good flow through the bed	Sand, gravel, long-term electron donors listed above , crushed rock, fractured nut shells
Solid surface sites for microbial attachment and retention	Provided by all above components; decreases as organic solids degrade
* Some of these products may cause incidental contamination and should be comprehensively tested before use.	

The previous description assumes a solid substrate system; one variant on a BCR is the active addition of a liquid organic substrate (see text box below). In this type of BCR, the media might simply be gravel or local trap rock with an initial inoculum of organisms, as the liquid organic feed rate is subject to control.

Advantages of Liquid Substrate Bioreactors

Advantages of Ethanol

In traditional, or first generation bioreactors a simple flow-through design is generally used, in which MIW is gravity fed into a pond or tank that contains an organic substrate such as manure, compost or wood chips. The MIW flows through the substrate and is released as water containing lower concentrations of metals with elevated alkalinity. The substrate acts as a physical framework for microbial attachment. In addition, the substrate serves as a carbon source for growth and maintenance of sulfate-reducing bacteria

These first generation bioreactors suffer from three inherent deficiencies that have been overcome by use of a liquid substrate (alcohol-enhanced) bioreactor. These deficiencies and the corresponding effects are:

1. *A first generation bioreactor decreases in permeability with time because the physical matrix is also the carbon and energy source. Over time the matrix is degraded, metals precipitate, a biofilm forms, the pore spaces decrease, and flow is restricted. In an alcohol-enhanced bioreactor, the matrix consists of nonreactive material, which maintains permeability over time as precipitated metals are removed during regular flushing of the matrix.*

2. *The rate of the sulfate reduction in a first generation bioreactor is difficult to control over long periods of time because the reactant substrate must be added in excess to allow for multi-year life. The mix of organic compounds is usually complex and the rate of oxidation of these compounds varies. As the preferred energy sources are depleted alternative compounds that are consumed at a slower rate dominate which results in a decreased rate of sulfate reduction over time.*

   In an alcohol enhanced bioreactor, an alcohol, (which is oxidized directly by SRB) is added at a stoichiometric concentration with the sulfate being reduced. Thus the rate of sulfate-reduction can be controlled for extended periods of time.

3. *Because the carbon source that is supplied is directly oxidized by the SRB, much higher rates of sulfate-reduction can be observed. Rates as high as 0.96 mol sulfate/(m3-day) have been observed in a methanol enhanced manure matrix reactor and rates as high as 0.56 mol sulfate/(m3-day) have been observed in a porous compost free reactor. Both were observed in the field at the Leviathan mine.*

Disadvantages of Compost-free Bioreactors

Although alcohol enhanced bioreactors have the advantages stated above, they also have some disadvantages. Since an alcohol must be continuously added, provisions must be included for storage, resupply and administering the organic liquid. In addition, these bioreactors typically require a higher initial capital investment due to the complexity of the flushing and fluid delivery systems. There is also an additional cost of continuously adding ethanol. However these costs are minimal (< $ 1.50/ 1000 gallons of MIW treated) compared to the cost of replacement of a spent substrate every 2-3 years. (Filipek et al. 2003)

Liquid substrates may be selected for design reasons that include the following:

- to start up a BCR
- to stimulate microbial activity in an upset BCR
- if space constraints limit the size of the BCR (such as in urbanized areas)
- if a low-cost liquid-phase substrate is available

Liquid substrates can be locally available waste materials, such as dairy whey, or commercially available products.

Selection of a liquid phase substrate or a combination of liquid substrates is typically based on the following factors: the ease with which the substrate can be degraded by SRB, cost, and ease of handling. Typically, liquid-phase substrates are simple organic compounds that are more easily degraded by SRBs than solid-phase substrates thereby increasing reaction rates within the BCR and reducing the size of the BCR required for a given flow and influent composition. Acids such as citric and acetic, or salts of the conjugate bases, may be selected for liquid-phase substrate BCR. Common antifreeze components ethylene and propylene glycol could also be tested. In addition, low molecular weight alcohols such as ethanol and methanol are candidates for liquid-phase substrate applications (see Section B.8). Low molecular weight acids, such as those found in the microbial citric acid cycle, are typically more easily degradable than alcohols to various classes of microorganisms. As a result, organic acids may be selected to support BCR start-up. Liquid substrate BCRs usually require more maintenance than solid substrate BCRs due to the need to store and replenish the liquid substrate. Control of the liquid substrate feed rate is important, although concerns over the potential for clogging of liquid substrate BCR from rapid growth of biomass are offset with adequate design and constraint of the substrate loading rate. Liquid substrates can also be used to decrease the start-up time of a solid substrate BCR with careful selection of the substrate type. An initial substrate screen can identify local sources for materials.

Selection of a liquid-phase substrate may also depend on whether the BCR design targets a diverse community of microorganisms, such as wood (cellulose) degraders, or targets growth of a specific

microbial population (as is typical for a BCR with no solid-phase substrate). This design decision is often based on the composition and ratios of the dominant electron acceptors in the BCR influent and the available space for the BCR. The potential for variability in the chemical composition of the BCR influent may also influence the selection of a liquid-phase substrate and associated feed equipment (see Section B.8).

3.1.1 Local Availability

In general, the substrate tends to be made of locally available materials. Other locally available materials that have been evaluated as potential treatment media include: biosolids, sugar beet waste, sugar cane waste, rice straw, peat, municipal solid waste compost, yard compost, paper (in limited amounts), and seaweed. The labile organic content of peat is relatively low, and while it might be used in other parts of a passive treatment system as an adsorbent, its use as BCR substrate is not recommended. Some wood species have elevated amounts of TDS (for instance, mesquite) that can be extracted by percolating mine water in a BCR. Wood and or straw derived from trees or plants grown near historical smelting sites could contain elevated levels of arsenic extracted from the soils. Bench testing is highly recommended for all sites in particular when new organic source materials are being considered

In general, numerous liquid-phase substrates could also be used in a BCR (such as alcohol). Liquid substrates can also be locally available waste materials, such as dairy whey, or commercially available products. Commercially available products used for BCR have included food-grade acetic acid, citric acid, methanol, ethanol, alcohol, calcium magnesium acetate, ethylene, propylene glycol, and sodium benzoate.

Resources for finding locally available substrate include the following:

- Search general internet search engines, electronic business pages, and phone books.
- Base searches on distance from site (25 to 50 miles).
- Search based on the key words for various materials presented in Table 3-2 below:

Table 3-2. Keywords for substrate materials

Function	Material
Long term carbon	Wood chips, sawdust, walnut shells, mushroom compost, other composts, chitin products,
Rice hulls	
Short term carbon	Manure, hay, straw, chitin products, yard waste, brewery waste, beet pulp, dairy whey, acetic acid, ethanol
Microbial inoculum	Ruminant manure, other BCRs, POTW sludge, sludge pond or slime dams at mining sites, or septic system products
Alkalinity	Limestone, seashells, fly ash, cement kiln dust
Permeability	Sand, gravel, wood chips, nut shells, crushed rock
Microbial attachment	Provided by all materials; in liquid reactors provided by rock matrix

Some materials are not likely to be locally available but may be cost-competitive. For example, the chitin used in BCRs to date is Chitorem® from JRW Bioremediation. When vendors are identified, contact them to determine what products are available, quantities in stock, rough unit prices, whether they have loading machinery (large orders), and who generally buys their materials (the competition). For composted materials, be aware that the quality of the compost stream from local governments may vary significantly over time as citizens vary what they provide for collection.

3.1.2 Substrate Composition

The initial substrate screen described in the previous section should result in a list of potential substrate candidates. The proportions and effectiveness of those candidates is then tested at laboratory-scale. Because laboratory-scale testing is relatively inexpensive, a variety of mixes should be considered. Each mix should be based on the roles shown in Table 1-1. Many of these mixes use a small percentage (2-10% total) of inoculum and short-term electron donor. Bulky items are generally not less than 10% of the substrate, so that hydraulic conductivity is not exceedingly low. The proportion of long-term organic source and alkalinity relies on judgment based on the MIW composition and expected pretreatment. For instance, BCR for strongly acidic MIW pretreated to pH 4.5 requires more alkalinity than a BCR treating MIW at pH 7.6.

3.2 Bulk Density Testing of Individual Components and Mixtures

Bulk density of all the materials used in a BCR should be determined to order the proper amounts for bench, pilot, and large scale systems. The objective of this test is to collect bulk density data that can be used to order the correct amounts of organic and inorganic materials for bench, pilot, and large scale sulfate reducing bioreactors.

3.2.1 Recommended Equipment

The equipment should be in good working order and calibrated to the extent practical. A bathroom scale, portable luggage scales, or more accurate digital laboratory scales can also be used for this test, along with a 20-liter/gallon capacity bucket or other suitable container for measuring volume. The scale used should have an accuracy of at least +/- 0.25 kg.

The 20-liter capacity bucket should be made of durable plastic with a handle. Calibration marks every 4 liters should be placed inside and outside the bucket, and determined by filling the bucket with water until the net weights are reached: 4, 8, 12, 16, and 20 kg (see Figure 3-1).

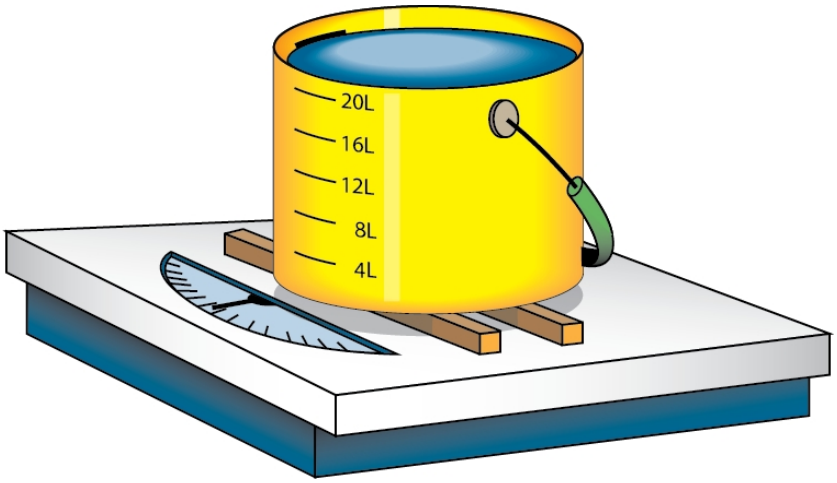


Figure 3-1. Schematic of a 20-liter bucket assembly to measure testing components.

3.2.2 Procedures

The procedure is somewhat crude but is typically accurate enough for large-scale material procurement purposes. Other variations of volumes and weights could be used, but should be conducted with the general procedures provided below:

1. First determine the weight of the empty bucket (tare weight). Then add the material.
2. To establish individual material density, the bucket should be filled with the material being tested to the 20 liter mark and compacted by stepping on the material with the weight of a typical person (90 kg). Apply the same level of compaction to all materials being tested so that the values derived are comparable. After the first compaction, top off the bucket to the 20 liter mark and weigh on the scale. Gently move or jostle the bucket to assure the reading on the dial is repeatable (approximately).
3. Record the tare weight of the bucket, the calibration volume (20 liters in this case), and the gross weight for subsequent calculations. Repeat as needed for all materials. If some materials tend to stick to the sides of the bucket, rinse the bucket with fresh water and dry with a towel.
4. Hay or straw bales are a special case. These are typically weighed intact directly on the scale. They may need to be stood on end and carefully balanced in the center of the scale pad. The bale volume is determined by direct measurement (L x W x H). If possible, several straw bales should be measured to obtain an average bale weight. An example worksheet for the calculations is provided in Table 3-2.
5. To establish a mixed substrate density, begin by using the proportions provided for the various organic and inorganic materials, then weigh out enough of each to prepare at least 25 to 30 liters of mixture. First fill the 20-liter bucket to the 16 liter level (80% full) with the most abundant individual material in the mixture, such as the wood chips. Obtain the weight of this volume and then weigh out the remainder of the mixture components proportionately.

6. Mix the components together thoroughly until relative homogeneity is achieved. It may be necessary to "shred" the hay or straw to short stem lengths for acceptable mixing for this small amount of volume. Measure the weight of the mixture. Using the same compaction practice as previously described, measure the volume of the mixture so that the bulk density and the "shrinkage" values can be estimated.
7. Shrinkage is estimated by dividing the volume of the mixture by the sum of the volumes of the individual components. For example, if 24 liters of the mixture (compacted) resulted from the combination of 16 liters of wood chips, 3 liters of manure, 5 liters of seashells, 3 liters of grape skins, and 6 liters of hay, the shrinkage factor would be:

$$24 / (16+3+5+3+6) = 0.73 \text{ or } 73\%$$

Table 3-3. Spreadsheet/checklist calculation table example

Bulk Density Field Estimates							
Bench Passive Treatment Systems							
All items in blue are input data							
Limestone	Lbs			Manure	lbs		
Volume		4.00	liters	Volume		5.00	liters
Gross Weight	15	6.8	Kg	Gross Weight	6.5	3.0	Kg
Tare bucket		0.50	Kg	Tare bucket		0.50	Kg
Net Weight		6.32	Kg	Net Weight		2.45	Kg
Bulk Density		1.58	Kg/liter	Bulk Density		0.49	Kg/liter
		98.4	pcf			30.58	pcf
Wood chips				Pixie Dust			
Volume		4.00	liters	Volume		4.00	liters
Gross Weight	1.5	0.68	Kg	Gross Weight	2	0.91	Kg
Tare bucket		0.50	Kg	Tare bucket		0.50	Kg
Net Weight		0.18	Kg	Net Weight		0.41	Kg
Bulk Density		0.05	Kg/liter	Bulk Density		0.10	Kg/liter
		2.8	pcf			6.4	pcf
Hay bale							
Volume		153.40	liters	Update with [control S]			
Gross Weight	81	36.8	Kg				
Tare bucket		0.00	Kg				
Net Weight		36.82	Kg				
Bulk Density		0.24	Kg/liter				
		14.9	pcf				
HAY BALE DATA							
1.31	kg per linear inch						
36.8	Avg kg of hay bale						
9,360	Cubic inches						
153.4	liters						

3.3 Proof of Principle Testing

The first step of proof of principle tests indicate whether a BCR or a given substrate inoculum or mixture of materials will be effective in principle in removing contaminants. This step is divided into two phases:

Phase 1 – Screening substrate material: Gather equipment and substrate inoculum candidate materials (see description above) and screen candidate materials

Phase 2 – Bottle testing: Set up bottle cultures, with substrate candidates and MIW combined, and monitor chemical and physical changes in the culture bottles for four to six weeks.

3.3.1 Phase 1 - Screening Substrate Material

Some initial substrate characterization is necessary to determine if some substrate materials can function as stand-alone BCR components or whether they need to be combined with other components (such as crushed limestone sand or liquid-phase substrates or conditioners) to provide BCR treatment. A screening process has been developed for initial substrate characterization. The equipment and materials needed to conduct the screening include:

- hand-held conductivity meter with calibration solution
- hand held oxidation-reduction potential (ORP) meter with calibration solution
- pH meter with calibration solutions
- disposable, re-sealable plastic bags (100mm x 100mm) to hold the materials to be tested
- digital camera

3.3.1.1 Procedure

1. Prepare a paste consisting of 1 part (by volume or weight) solid substrate material (for example: wood chips, sawdust, manure, or hay) and 2 parts distilled water. Mix thoroughly and allow to stand for one hour. The paste can be mixed in sandwich-sized heavy-duty plastic bags, which can be subsequently discarded. Be sure to label the bags with an indelible marker.
2. Usually about 50 grams of solid material and 100 ml of distilled or de-ionized water will suffice for this test. If the solid material absorbs water excessively, additional moisture may be needed. If necessary add water, record how much is added, and wait another hour before taking measurements as described below.
3. Test the paste for pH, conductivity, and ORP. To obtain a measurement, collect the paste into a corner of the plastic bag and carefully insert the pH/conductivity/ORP probe(s) into the paste so as to avoid damage to the probe. Make sure the tip of the probe is under water or in good contact with the paste.
4. For best results, measure the pH values of the samples first, then measure the conductivity of the pastes, followed by the ORP values. The probes should be rinsed with distilled water after each measurement. If an ORP probe is not available, the pH and conductivity values will suffice. Record all values. Photograph the individual substrate materials close up with a digital camera (equipped with a macro lens if possible) with a ruler or scale in the photo.

3.3.1.2 Data Interpretation

To be a good stand-alone substrate candidate, the paste should ideally have:

- *Near neutral pH*. This value demonstrates that the substrate alkalinity is able to dissolve into the distilled or de-ionized water.
- *Conductivity value of at least 2000 µmhos/cm*. This value indicates the substrate alkalinity is able to dissolve into the distilled or de-ionized water and that ion exchange for short-term adsorptive removal is possible.
- *ORP value of less than 200 mV*. To convert ORP readings taken under these conditions to Eh, simply add 200 mV to the ORP voltage (see "Measuring ORP" from YSI Environmentalhttp://www.ysi.com/media/pdfs/T608-Measuring-ORP-on-YSI-6-Series-Sondes-Tips-Cautions-and-Limitations.pdf). This value shows that biological activity has used the oxygen that was in the water, confirming organisms and short-term food are present. An ORP meter reading should be corrected to Eh, since this hydrogen-referenced value is standard when describing redox in biology.

3.3.2 Phase 2 - Bottle Testing

The goal of Phase 2 is to identify which substrate materials or combinations of materials may provide the better recipes for the BCR design. Identifying more than one good recipe allows reasonable substitutions during construction if certain materials are in short supply or become too expensive. At the end of the 4- to 6-week bottle test, three of the most indicative culture bottles and two duplicates should be analyzed in detail for metals removal and other parameters. Indicative BCR bottles are also analyzed for sulfide to show that sulfate reduction (and concomitant sulfide production) occurred. Composted animal manure obtained from commercial gardening centers provide a benchmark source of BCR bacteria culture inoculants.

Dozens of test bottles with duplicates can be monitored during the course of the proof of principle test. This extensive monitoring is the key advantage to a proof of principle test using culture bottles. By using small amounts of substrate and MIW, and keeping to a simple experimental design, the test can assess a greater number of removal possibilities and better indicate appropriate substrate and aqueous conditions. Because of cost and time considerations, test only those substrate that are most likely to work based on experience and material availability.

Optimization of the substrate inoculum for maximum contaminant removal is the final step, and data to support this step is developed through dynamic bench scale testing further described in Section 3.4. Success in the proof of principle test is defined by a measurable (typically 50 percent) decrease in contaminant concentration over the course of the experiment, rather than a decrease compared to a water quality standard.

3.3.2.1 MIW Collection Procedures

Begin the Phase 2 bottle testing process by obtaining at least 8 liters of the mine drainage water to be treated. During the collection process, observe the following procedures:

- Collect unpreserved, whole (unfiltered) MIW.
- Check and record pH and other field parameters.
- Rinse each sample collection container once with a small amount of MIW.
- Transport the sampled raw MIW for proof of principle testing immediately to the lab treatability testing facility and keep on ice or refrigerated.
- Fill appropriate sample collection bottles, add preservatives as appropriate, and ship samples to the analytical laboratory for testing.

The MIW sample collection bottles used should filled completely and capped to prevent oxygen intrusion. Include appropriate duplicates of the MIW so that intermediate and confirmatory tests can be performed during the course of the experiment. Assess preliminary results of the analyses after two weeks of testing, and weekly thereafter as results are available.

3.3.2.2 Bottle Setup Methods

From the eight liters of MIW, two 250 mL aliquots should be taken and properly preserved. These aliquots are set aside to be available for analysis at a future date if necessary. Two additional 250 mL aliquots should be analyzed for pH, Eh conductivity, alkalinity, and a full suite of metals and anions, including sulfate. A Hach kit can be used to measure sulfide (which is expected to be low in typical MIW, for instance, less than 0.1 mg/L sulfide).

The quantity of MIW used in proof of principle tests are project specific, but at a minimum should provide sufficient MIW volume to allow for collection of sample aliquots throughout the test or at the end of the test, including laboratory analysis of metals and other parameters. Proof of principle bottles should contain substrate (at as-received moisture) weighed into the bottle. The unfiltered MIW should be added at the rate of about 250 ml for experiments with 100 g of substrate plus 30 g of inoculum. Minor amounts of distilled water can be added to minimize head space.

1. Shake the bottle to mix the inoculum with the substrate materials.
2. Before capping and sealing, measure the pH, Eh and conductivity of the solutions.
3. After measurements, all BCR test bottles may be purged with nitrogen or argon gas for approximately 1 minute to remove dissolved oxygen – this is optional.
4. The samples should be incubated at room temperature, 25 to 30° C.
5. For BCR tests, the electron acceptors are NO_3 (denitrifiers), SO_4^{2-} (sulfate reducers) and CO_2 (methanogens). These compounds produce gases of N_2, H_2S, and CH_4, respectively (some more than others). Therefore gas appearance is expected but does not confirm sulfate reduction.
6. The progress of sulfate reduction can be followed by observing the color of the sediment in the bottles, which changes from brown to black. Test for free sulfide using a Hach kit.
7. The Eh (redox) of the solutions also indicates that sulfate reduction is probably occurring; sulfate reduction is expected to occur in reducing conditions of less than -150 to -200 mV.

3.3.2.3 Observation Schedule

Every seven days, inspect the anaerobic bottles for bacterial activity: gas evolution, bubbles, color changes, and the smell of rotten eggs (sulfide). In addition:

1. At 14, 21, and 28 days, test for sulfide, metal of concern, Eh, and pH of accessible culture solutions.
2. Between 21 and 28 days, photograph representative culture bottles.
3. At 28 days the experiment is considered finished, unless no significant microbial activity has taken place. Measure Eh, pH, and conductivity of all the solutions.
4. Based on Eh, pH and appearance, the 'best' of the BCR bottles should be analyzed for the same suite of parameters as the original suite.

3.3.2.4 Interpretation of Results

The bottle tests demonstrate the production of alkalinity and the amount of sulfate reduction and sulfide produced by the substrate mixes that are tested. An ideal result is significant sulfate reduction (100 mg/L or more) and sulfide production (5 mg/L or more), pH near neutral, and Eh below -200 mV. The presence of sulfide should be accompanied by very low concentrations of dissolved metals other than manganese, which is not well removed in a sulfate reducing system (see Appendix A.1.4).

3.4 Bench-Scale Testing

This phase of testing for the flow through system is typically performed in the controlled environment of a laboratory but can be conducted in the field. It is most appropriate for evaluating the dynamic response of different mixtures of organic substrates, system configurations, or metal loading rates. This level of testing should be relatively inexpensive to set up; most of the cost should be allocated to sampling and analysis. To keep costs low, bench-scale test units can be constructed with off-the-shelf items such as trash cans and wading pools. Once the range of dynamic variables has been narrowed, on-site pilot testing can proceed. Sampling should be conducted as frequently as possible throughout the course of testing (weekly or bi-weekly) in order to develop time-based correlations of sulfate reduction and metal removal. Flow-through tests conducted with plastic 55-gallon barrels are also commonly used for field tests.

Bench testing (dynamic), not less than three months should include the following:

- The test should show how the variability in site conditions can affect the outcome of the test.
- The test should be long enough to demonstrate sulfate reduction and distinguish it from adsorption.
- The test is a small-scale field or in-house test using site water.
- Test flow conditions (for instance, continuous or pulsed) likely to be encountered at the site.
- Use caution when transporting MIW as precipitation may occur, thus altering the chemistry of the water during transport.

- The test may use MIW surrogates in some cases when pretreatment is expected to produce a specific MIW condition but cannot be implemented, or when the site is inaccessible during the period of testing (for instance due to high elevation or winter weather).
- Test the media from the proof of principle test under flow through conditions.
- Test posttreatment methods after the BCR testing unit(s)
- Test the acid, metal, and hydraulic loading range expected in the field.
- Test acceptable flow through characteristics of the substrate.
- Identify any pretreatment that may be required to avoid substrate plugging.

The ideal result of the bench-scale testing is to determine the best combination of substrate mixture and loading rate deemed to yield acceptable water quality. The results should include an estimate of the sulfate reduction rate per volume of substrate based on observed sulfate removal. The observed metal removal efficiency (MRE) should also be determined. It is important that the rate of sulfate reduction is evaluated for an extended time to determine the long-term rate as opposed to the initial rate. Eger and Wagner (2001) noted in a barrel-scale study that the observed rate of sulfate reduction was initially as high as 300 mmol/m^3/d but declined to 20 mmol/m^3/d, and that rates were highly variable in part due to seasonal operation.

To determine preliminary K (hydraulic conductivity) values of the substrate, a falling head permeability test may be conducted.

3.5 Field Pilot-Scale Testing

This phase of testing is performed at the site, on the actual MIW. Information gathered during these tests should provide an accurate operating cost estimate as well as final capital cost data. The field pilot-scale test is a scale-up based on the small-scale bench test including any anticipated pre- and posttreatment units. The size of the pilot-scale depends on the site and is a balance of cost and demonstrating behavior at full-scale. In some cases pilot-scale units have been constructed using parts of above-ground plastic swimming pools, with footprint dimensions in the range of 3 to 6 meters in diameter. Field pilot-scale tests can also be conducted in ground using lined ponds. Pilot-scale substrate mixing and placement often is manually intensive, requiring buckets for measurement and concrete mixers, or using Bobcat-size equipment and garden tillers for measurement and mixing.

In order to minimize earth work, pilot-scale tests frequently are bermed instead of completely in-ground and may require pumps to feed MIW to the system. Because the range of sites at which MIW is found, this section does not specify design and operation. Instead, see descriptions of pilot-scale systems in the literature (Thomas and Romanek 2002; Reisinger, R.W. and J.J. Gusek 1998; Gusek et al. 2000; Reisman et al. 2008; Gusek et al. 2008; Faulkner et al. 2007; Gusek and Schueck 2004).

The duration of the pilot-scale test in the field should be sufficient to demonstrate treatment effect during any anticipated variations in MIW character. A duration should be set that encompass the seasonal and other variability as described in Section 2.1 to the extent practical. In many cases, at

least a year is required to see the full range of variability in MIW at a site and also to monitor performance at the coldest and hottest conditions.

Data collection during the pilot test should include the following:

- Initial substrate bulk density and volatile suspended solids. In addition the substrate could be tested for cation exchange capacity (CEC), reflecting adsorption capacity, and cellulosic content by sequential extraction as described in Hall (2000).
- Water quality parameters of interest for the influent and the effluent. For many sites these parameters include pH, dissolved oxygen (DO), Eh, conductivity, alkalinity, acidity, temperature, ammonia, nitrate/nitrite, sulfate, sulfide, and concentration of metals of concern.
- In situ Eh within the reactor column, DO, and pH
- Ending vertical profile of Eh and pH, reflecting degree of exhaustion of alkalinity and organic, substrate bulk density and volatile suspended solids. If initial substrate was tested for CEC and cellulose, also test final content.

If the field pilot study does not meet the necessary discharge standards, another treatment technology should be considered or additional pre- or posttreatment added. In most cases pretreatment at pilot-scale will be part of the study along with posttreatment methods to oxidize the BCR effluent. It is also important to determine the sludge characteristics, assuming sludge is produced, during this phase. Will the sludge be hazardous or nonhazardous? Can the treatment sludge be disposed of on the mine site? Sludge management and organic substrate replacement may comprise the principle operating costs of a BCR treatment system.

Table 3-4. BCR substrate functions and common components

Role	Examples of Components*
Source of long-term electron donor (food)	Wood chips, sawdust, walnut shells, mushroom compost, other composts, chitin, rice hulls
Source of short-term electron donor for start-up (fast food)	Manure, hay, straw, chitin, yard waste, brewery waste, beet pulp, corncobs, dairy whey, acetic acid, ethanol
Source of microbial inoculum	Ruminant manure, other BCRs, POTW sludge, sludge pond or slime dams at mining sites, or septic system products such as RidX®, manure from browsing animal has a higher ability to break down cellulose
Source of alkalinity	Limestone, seashells, fly ash, cement kiln dust
Bulk to allow for good flow through the bed	Sand, gravel, long-term electron donors listed above , crushed rock, fractured nut shells
Solid surface sites for microbial attachment and retention	Provided by all above components; decreases as organic solids degrade
* Certain of these products may cause incidental contamination and should be comprehensively tested before use.	

At the conclusion of field pilot testing three equally important aspects affect full-scale passive treatment system design: biology, MIW chemistry, and precipitate retention. The bench and pilot test

results should have yielded the conditions necessary to establish the proper chemistry or dominant geo-ecosystem in a given treatment unit to develop stable chemical precipitates. However, constructing an ideal biochemical environment is a wasted effort if the metal precipitates formed are flushed out of the system because of inefficient retention. Among other factors, this aspect of a proper system design is influenced by the grain-size distribution and compacted density of organic substrates, the settling and flocculating characteristics of the precipitates, and the retention times of the settling units.

Brewer Gold Mine

At the Brewer Gold Mine in South Carolina (Appendix B.6) two anaerobic pilot cells were constructed with a mixture of turkey manure, sawdust, phosphate rock, and cow manure inoculum. The design was based on 0.3 moles of sulfate reduced/day/m of substrate. The influent concentration fluctuates due to rainfall. The pad surface was immediately vegetated unintentionally causing erratic redox/Eh, which paralleled iron removal efficiencies. High redox resulted in remobilization of copper and aluminum, which prompted the elimination of the vegetation. The system returned to an anaerobic state after vegetation removal and the percent removal of iron, aluminum and copper stabilized in a range of 95% - 100%.

4.0 DESIGN

This section presents a BCR design strategy, including discussion of general design considerations, full-scale system scale-up, design components, outfall, hydraulics, materials, and site control. General submittal recommendations for design of BCR systems is also presented. The contaminants of concern may be removed aerobically or anaerobically (see Figure 2-2), but the focus in BCR design is on sulfide production in the anaerobic zone.

As noted in Section 3.0, inherent dangers exist in any cookbook design approach because the conditions on-site may lie outside the range of conditions that were originally used to develop the standardized design criteria. MIW problems present a wide range of possible conditions, and a specific site could fall outside the guidance provided here. Use professional judgment when considering whether the advice provided in this document is appropriate for a given particular site.

A wide range of technical knowledge (presented in Section 1.0 and the appendices) and skills are required to design a BCR. The design effort usually requires a team that includes capabilities in geochemistry, chemical analyses, engineering, hydraulics and hydrology, fieldwork, soils and foundations, construction estimation, and microbiology. Team members should communicate effectively.

The difference between equilibrium and kinetics (rate) as described in Section 1.3 is another concern for design. Equilibrium usually predicts the formation of insoluble metal sulfides. Some metals, however, may be kinetically trapped by adsorption to natural organic matter and therefore not able to react with sulfide. If the organic matter is colloidal, the attached metal flows through the system untreated. Thus equilibrium expectations can be violated due to kinetic rates. Some BCR systems have much higher initial effluent concentrations of total metals than the dissolved metal forms, and many believe that colloidal materials carry much of the metal found in a BCR effluent. Due to both equilibrium and kinetic realities, BCRs may not fully treat each metal in MIW or completely remove other constituents. In some cases BCRs can discharge elevated levels of other constituents such as sulfides, nutrients and organic material. This can be a concern particularly during the start-up phase of a BCR, but can also be a concern during operation of the BCR (see Section 4.2).

4.1 BCR Design Evolution

BCRs were originally inspired by natural wetlands and therefore were considered a form of constructed wetland. Treatment wetlands are designed largely on areal loading and residence time, and a similar design approach was initially used for wetland treatment of MIW. Kleinman (1986) suggested 5 m^2 of wetland per L/min of mine drainage, a loading rate of 0.29 m^3/m^2-d. This value was subsequently revised to 15 m^2 per L/min (0.096 m^3/m^2-d) (Girts et al. 1987).

In contrast to a hydraulic loading rate, Brodie et al. (1988) proposed an area per mass loading of 0.75 m^2 per mg Fe/min. Dietz et al. (1994) suggested a similar areal design on acidity, based on 6

g/d/m^2. However, the failure of a single areal rate to describe the performance or design of a unit had become apparent. For example, Hedin and Nairn (1990) found no clear relationship between iron areal load and removal for three different wetlands.

The idea of a volumetric rate of sulfide production (or sulfate reduction) has also been put forward to describe BCR design; however the initial rate of sulfide production is in excess of the long term design rate. Recent design developments have incorporate the blending of effluent and influent water to utilize excess hydrogen sulfide and sequential use of the organic substrate. In this case, design should be based on a total sulfide production rate in mol/d greater than the expected total metal load in mol/d. McIntire and Edenborn (1990) reported rates comparable to coastal marine sediments, with 2.0 to 600 mmol S $m^{-3}d^{-1}$ measured at the Friendship Hill (PA) system (see Friendship Hill Case Study, ITRC Mining Waste Treatment Technology Selection) treating water with pH of around 3. Wildeman's group at Colorado School of Mines similarly offered 300 mmol S $m^{-3}d^{-1}$ as a conservative estimate based on the Big Five wetland (Machemer et al. 1993). Lower values, however, have been reported; Mueller et al. (1996) determined sulfate removal rates of 128 to 547 mmol S $m^{-3}d^{-1}$. The studies at the Big Five tunnel also showed that permeability of the substrate was a critical design factor for successful constructed wetland operation.

The first published attempt to model geochemical and biogeochemical processes in a BCR seems to have been by Klusman around 1990 (Klusman et al. 1993). MINTEQA2 code was modified to calculate rates, including an assumed sulfide generation rate, and model plug flow by doing time steps on a batch case. Although capable of matching effluent concentrations from several operating wetlands (BCRs) to some degree, the authors noted that inadequate experimental data existed to confirm model results against wetland realities.

In addition to the question of design values, specifics of design have been the subject of doubts. Is the direction of flow better as vertical down, vertical up, or horizontal? Which redox condition, aerobic or anaerobic, results in better treatment? Are plants important? On the subject of flow direction and redox, Dietz and Stidinger (1996) studied paired surface-flow (horizontal flow with open water) and vertical flow (downflow) wetlands in series and parallel and found subsurface flow performance to be significantly better for alkalinity impact, acidity reduction, and iron and aluminum removal. Manganese and calcium were significantly higher in the vertical flow effluent, the latter presumably due to calcium carbonate dissolution related to acid neutralization. They speculated that vertical flow had improved contact between substrates (both mushroom compost and limestone gravel) and the impacted water. Although subsurface flow is now recognized as desirable for removal by sulfide precipitation, the direction of flow has not been shown to affect removal. That noted, most units are vertical downflow for simplicity.

The role of plants also came into question in the early 1990s. If removal was by sulfide precipitation and wetland plants aerate the subsurface, perhaps plants were not desirable. Not having plants would also decrease concerns over bioaccumulation. On the other hand, plants increase microbial counts in the subsurface and as the senescent mass decays, replenish the bed organic matter. As noted in Beining and Otte (1997), a natural wetland in Wicklow, Ireland, has treated MIW for more than a century and, based on plant growth, is calculated to continue treatment for at least

another century. A powerful counter-example of the potential role of plants was the shallow Brewer Pad 5 system (see Section B.6) operated from 1993 to 1995. When vegetation was present, the system was aerobic and copper removal efficiency was poor. A deeper unit would allow for increased anaerobic volume where the plant roots may not go. Although it is not clear whether plants are a net benefit, current BCR designs often exclude plants through placement of a geo-textile cap, a thick gravel layer, or a free water surface with a depth of one meter.

4.2 BCR Design Considerations and Criteria

Following Figure 2-1 and the testing protocols in Section 3.0, the design team performs the following tasks prior to design:

1. Characterize the MIW influent quality and flow.
2. Characterize the site.
3. Identify the treatment goal, including applicable rules and regulations.
4. Consider and possibly select pretreatment, based on the material presented in Section 2.4.
5. Estimate the amount of sulfate reduction sufficient to precipitate metals as sulfides.
6. Perform bench-scale tests (Section 3.4) in the laboratory, on site, or both to determine:
    - loading range, substrate thickness, and residence time
    - substrate mixture
    - substrate degradation rate
    - metal removal efficiency (MRE)
7. Perform pilot-scale testing at the site for an extended period, which establishes performance at the specified loading rate and with the specified substrate mixture.
8. Evaluate effluent quality for compliance with applicable discharge standards including water quality parameters such as nitrate/nitrite, ammonia, phosphorus, biological oxygen demand (BOD).

Given the determination of the various design variables and constraints as described above, the following subsections describe in greater detail some additional design considerations, including influent chemistry concerns, site conditions (including discussion of BCR covers), temperature effects, flow variability, and parallel treatment trains.

4.2.1 Influent Chemistry Concerns

Some characteristics and constituents of MIW can adversely affect BCR performance, so MIW pretreatment is usually necessary. For example, iron and aluminum both can produce sludges, which rapidly plug BCR pores. Specific pretreatment technologies are reviewed in Section 2.5 and more detail is available in ITRC MW-1 (ITRC 2010). In addition to the characteristics and constituents listed below, Appendix A Physical, Chemical, and Biological Reactions in a Biochemical Reactor includes additional metals that are less common in MIW influent but still important.

4.2.1.1 Iron

As MIW is neutralized and increases in ORP occur, iron oxyhydroxides and hydroxysulfates form (see A.1.2). These solids are dense and deposit readily on surfaces. The solids also tend to form a large volume of fluffy flocs that can rapidly plug a porous bed. If the MIW is acidic and contains more than 100 mg/L (Reisman et al. 2008) of iron, then the passive treatment system design must consider pretreatment to remove iron. At lower concentrations, iron still can be challenging and pretreatment is advisable. The Pourbaix Eh-pH diagram (Figure 1-2) shows that iron will almost always stay in solution below pH 2, and in highly reduced conditions remains as Fe^{2+} (aq) up to near-neutral pH. The decision tree in Figure 2-3 (link to Figure 2-3) shows that most often the pretreatment chosen in this case will be an open limestone channel (OLC) (Table 1-1) which raises both pH and Eh, causing iron oxyhydroxides to form, followed by a settling pond. In some cases a SAPS/RAPS may instead be used followed by aeration and settling or followed directly by the BCR unit.

As shown in Figure 2-3, an anoxic limestone drain (ALD; see ITRC MW-1 2010) is appropriate pretreatment for anoxic (dissolved oxygen (DO) less than 1 mg/L) MIW with modest amounts of ferric iron (Fe^{3+} less than 1 mg/L). After the ALD adds alkalinity, the water should flow to a settling basin to allow the iron oxyhydroxides to settle.

4.2.1.2 Aluminum

Aluminum hydroxides are a favored floc form for removing colloids from drinking waters. Aluminum hydroxides tend to be gelatinous, sticky sludges, which can rapidly plug a BCR and thus if aluminum is present around or above 10 mg/L (Reisman et al. 2008), pretreatment for this metal must be considered. Because aluminum removal requires a higher pH and thus more alkalinity to remove than iron (Section 1.3), the pretreatment most often used is a SAPS. Unfortunately, the sludge is formed atop and within the SAPS, and periodic flushing or substrate replacement may be required.

4.2.1.3 Mercury

If mercury (Hg) is present, the potential formation of methylmercury in the system can be a concern due to the reducing conditions. Although sulfate reducing conditions can capture Hg, the reducing conditions also allow the formation of highly toxic and mobile methylmercury (King et al. 2002). Sulfate reducing bacteria (SRB) have also been shown to promote methylmercury formation. If total mercury is analyzed, the analysis does not measure methylmercury and the effluent may be more toxic than anticipated due to the methylmercury content. If mercury is present testing for methylmercury should be incorporated.

4.2.1.4 Potentially Inhibitory Constituents

BCRs rely on the activity of microorganisms. If contaminant levels become too high, the microbial culture may be adversely impacted, especially by low pH or high levels of zinc or copper (50 and 5

mg/L, respectively (Ruhs et al. 2006); see Section B.1.4). If the MIW has high zinc or copper concentrations, pretreatment improves BCR performance. Another method to reduce high zinc and copper concentrations for treatment is to simply dilute the influent to a BCR with recycled effluent or with other water low in the metals.

For low pH, pretreatment may be required. Both cellulose degraders and SRB do not tolerate pH changes. Although SRB have been reported to thrive in acid conditions, these acid-tolerant SRB most likely are not the same organisms found in the manure used as a BCR component. Perhaps more importantly, cellulose degraders are believed to cease activity below pH 5 (Qu et al. 2005). The BCR influent therefore should not be too acidic (less than pH 4). The alkalinity in the BCR neutralizes pH to some extent, but also biological activity will affect pH. The formation of carbon dioxide from organics results in a shift in carbonate equilibrium, potentially lowering pH slightly as carbonic acid is formed. On the other hand, sulfate reduction results in sulfide, but the dominant species is likely to be HS^-, thus raising pH. The effluent pH from BCRs is often above 6.0 (USEPA 2001).

4.2.1.5 Oxygen and Nitrate

Microbial metabolism of oxygen and nitrate yield more energy to microorganisms than does sulfate reduction. Since many BCR treatment processes involve sulfide generated by SRBs, design of a BCR must consider oxygen and nitrate in the MIW that is treated by a BCR. If MIW contains oxygen and/or nitrate, an aerobic zone and nitrate-reducing zone must be maintained within the BCR. In this anoxic zone, nitrate consumes the refractory BOD and soluble organic substrate. Nitrate is reduced to nitrogen gas by denitrifying bacteria and consumes acidity. Nitrate treatment likely improves water quality although it also decreases BCR efficiency in terms of sulfate reduction and sulfide generation. A portion of the organic in the substrate is used for nitrate reduction, and thus the sulfate reduction rate is decreased and the BCR consumes more substrate per time increment. Rather than pretreatment, the testing phase described in Section 1.3 should include characterization of the anoxic layer depth by Eh and estimate the rate of depletion of organics due to oxygen and nitrate reduction (that is, increasing the BCR electron donor demand). The presence of oxygen or nitrate in the MIW may require additional media depth and volume in the BCR to maintain a comparable design sulfate reduction and sulfide generation design criteria.

4.2.2 Design for Site Conditions

Siting a BCR system depends primarily on the location of the source of MIW to treat. Most BCRs are located in remote, hard-to-access areas close to the source of the MIW. These settings are often also in steep terrain, which dictates the area available for the BCR; some BCR systems snake down a hillside like the switchbacks of a mountain road. Private land may abut the location, so consider the value of land purchase in site planning. Be creative when considering placement of a treatment system in steep and challenging terrain, and remember that construction may require added flat space. Maintenance personnel must have access to the site as well.

When the terrain is challenging, one design alternative is to pipe the MIW a significant distance to

a more suitable and accessible location. An advantage of steep terrain is that often the elevation difference between the location of the MIW and the BCR system allows gravity flow to feed the MIW through the BCR other treatment units, and to the receiving water.

In addition to topography, factors to consider while siting a BCR system include local precipitation, vegetation, type of soil at the water source and at the BCR system site, availability of materials and resources needed to build the BCR system, and neighbors. This guide does not address soil mechanics or the design of foundations, but the design team must also consider these issues. The availability of utilities during construction and operation is another site-related issue that must be considered if energy is required. If electricity is unavailable, solar or wind power and battery storage is a proven option for operating remote systems.

The site specific and general topography of the mine area should be evaluated during the siting of the location. Address the following questions regarding the mine area:

- Where will the MIW be captured and conveyed to the BCR system?
- Where will the BCR be located?
- Where will posttreatment units such as aerobic polishing cells (APCs) and final discharge (outfall) effluent be located
- How will equipment get to the site?

Establishing a good relationship with the local populace, if present, is important because happy neighbors are more likely to report observed problems at the site, and are less likely to protest. If present, the neighbors should be contacted early in the process and inform them of the reason for the proposed system, the expected construction issues and duration, the finished appearance, and frequency and magnitude of maintenance visits. BCRs treating large sulfate loads have the capacity to off gas low levels of hydrogen sulfide. Hydrogen sulfide odors can be intensely unpleasant and the human sense of smell becomes rapidly fatigued by exposure to these odors. Hydrogen sulfide is hazardous at low concentrations National Institute for Occupational Safety and Health-Recommended Exposure Limits (NIOSH REL) is 10 ppm for 10 minutes. If the site can generate significant hydrogen sulfide emissions, then site control is required and a provision for an odor control system such as oxidation, aeration, and biofilters should be included in the BCR design.

Because most MIW is found at active, abandoned, or reclaimed/closed mines, hazardous materials may also be present both above and below ground at a site, including below ground surface. The design should include controls for all known or suspected contaminants that workers may be exposed to during construction and maintenance of the BCR. One such control is to create an on-site repository or to find an off-site location to accept contaminated materials.

BCRs located in cold weather environments may need to be protected from freezing conditions. If the MIW chemistry allows (it has virtually no dissolved iron, aluminum, or suspended solids), then

the BCR can be fed from the bottom, similar to a **fen**[1]. Otherwise, the BCR could be buried. Some examples of buried BCRs can be found in the case studies:

- B.4 Ferris-Haggerty Mine Case Study (second pilot), Wyoming
- B.5 Fran Coal Mine Case Study, Pennsylvania (Gusek and Schueck 2004)
- B.9 Wheal Jane Mine Case Study, Cornwall, UK

BCR cover designs (downflow units) should consider the following:

1. What is the frost depth or expected snow cover at the site?
2. To minimize the geotechnical surcharge loading (and consolidation/compression) of the substrate, the cover fill material should be as light as possible. Wood chips have been used in this capacity, but see item 5 below.
3. For acidic MIW with dissolved iron and aluminum, the cover design should allow a zone of free standing water immediately above the substrate. This practice avoids the need for perforated pipes (which can plug) on top of the substrate.
4. If a high permeability zone of lightweight material is to be in direct contact with substrate, there should be limited suspended solids allowed in the BCR influent and the lightweight fill material should be inert with respect to altering the MIW influent chemistry (for example, limiting the possibility for precipitation reactions to occur).
5. If a high permeability zone of light weight organic material (such as wood chips) is in direct contact with substrate, any alternations in MIW chemistry should not cause precipitation of solids that could cause plugging.
6. Any geomembrane or geotextile materials used in the cover design should be covered with a soil layer that can be revegetated with native grasses.
7. Covers should include dedicated ports for sampling the substrate over time.

[1]Fens are peat-forming wetlands that receive nutrients from sources other than precipitation, usually from upslope sources through drainage from surrounding mineral soils and from groundwater movement. Fens differ from bogs because they are less acidic and have higher nutrient levels. They are therefore able to support a much more diverse plant and animal community. These systems are often covered by grasses, sedges, rushes, and wildflowers. Some fens are characterized by parallel ridges of vegetation separated by less productive hollows. The ridges of these patterned fens form perpendicular to the downslope direction of water movement. Over time, peat may build up and separate the fen from its groundwater supply. When this happens, the fen receives fewer nutrients and may become a bog. See EPA website: http://water.epa.gov/type/wetlands/fen.cfm

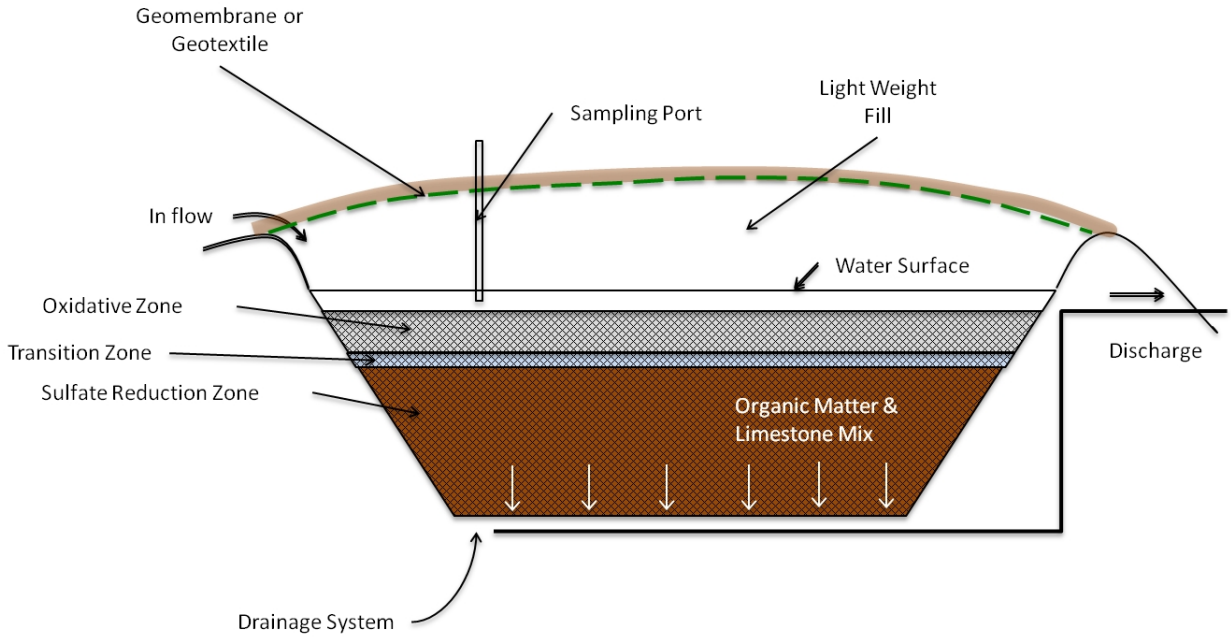


Figure 4-1. Schematic cross section of a cover design.

To develop a zone of free standing water, the septic system infiltration chambers or similar units can be placed directly atop the substrate and covered with lightweight fill. These chambers are usually made of plastic or concrete. If the MIW is very acidic, plastic chambers should be specified, because concrete chambers are subject to corrosive attack.

The cover design engineer should assess the need to include geomembrane or geotextile layers in the BCR cover. This determination is based on the climate and the grain size distributions of the various materials in the cover. These materials may be segmented in a way that allows their preservation, salvage, and re-use when the substrate is periodically exhumed and replaced.

The inclusion of a cover restricts BCR inspection and sampling of substrate materials for long-term consumption and degradation (see Section 6.0). Consequently, multiple sampling ports for collecting substrate materials should be included in the cover design. Some designers have included five "nests" of sampling ports in a square BCR as shown in Figure 4-2.

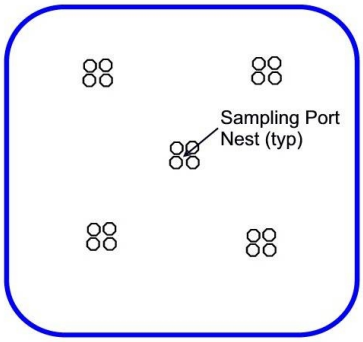


Figure 4-2. Schematic of five nest sampling ports in a square BCR.

If the surcharge load on the substrate is a concern, geotechnical testing should be conducted to evaluate how the substrate behaves in response to the extra weight. As the substrate ages, its ability to support the cover load may decrease and the cover surface may subside. Assuming uniform substrate degradation or consumption, the thinner zones of the substrate (around the perimeter of the BCR) may subside less than the center and the cover may lose its ability to shed rainfall or runoff. The project engineer should consider these differential settlement amounts in the cover design to minimize this effect.

4.2.3 Temperature Effects

Bench- and pilot-scale systems are more susceptible to ambient air temperatures than full-scale in-ground systems. The temperature in the test systems varies more and extends lower and higher than it does in the full-scale units. The effects of temperature on biological activity are also unpredictable. Although the biological rule of thumb is a doubling of activity for every 10°C, increase in temperature in the range 4 to 40°C, some systems have given higher volumetric rates of sulfate reduction as temperatures lowered (for instance, see Section B.4).

If the influent water is especially cold (less than 10°C), then it may be beneficial to incubate (during start-up) the BCR at higher temperatures (greater than 10°C, but less than 20°C) to mature the microbial population (see Ferris-Haggerty).

BCRs and their component parts can freeze, especially since mines are frequently located at high elevation. Heat lost from the water is equal to the heat gained by the air, assuming little heat transfer to the soil. The heat balance in a BCR is thus analogous to that for ponds:

$$Q(T_{in} - T) = hA(T - T_{air})$$

where:

Q = flow rate, m^3/s

T_{in} = influent water temperature (often the same as local groundwater) °C

T = mean temperature in the BCR °C

A = area exposed to air (footprint) m^2

h = heat transfer coefficient (m/s)

Although a value of k for open water (and measured in areas with minimal snowfall) has been reported (Mancini and Barnhart 1968), this equation may not be useful for most mine waters because of the low temperatures that occur at mining sites. For a given location's lowest sustained temperature (T_{air}), if T solves to zero or lower, an insulating layer of ice will form. The ice (and snow) insulation thus changes the value of k so that it decreases heat loss to the atmosphere and the value of k given above no longer applies.

4.2.4 Flow variability

As emphasized in Section 2.0 and during the testing process described in Section 3.0, the MIW flow characteristics should be observed over several seasons. To account for variability in flow, equalization and storage volume may be required. The design of such storage systems is described in numerous textbooks, but for MIWs, the designer should especially consider corrosivity and possible sludge generation during storage. The BCR must be designed to accommodate the highest hydraulic load, but also must include capacity to safely discharge overflow.

The preliminary data collected should include not only flow rates but also precipitation. The design team should analyze this data for correlations, because in many instances the volume of water to treat is directly related to the local precipitation of the area (with some delay period). In this case, prevention is preferable to treatment, and diverting stormwater away from the former mining site results in a smaller volume of water to treat. If a strong correlation between rain events and flow rates exists, the design may focus on decreasing the permeability of the collection area, increasing evapotranspiration, channeling storm water, and creating detention basins if required. In other cases, the former mining sites are influenced by not only the local precipitation, but also by groundwater that finds its way to the mining site. In this latter case, extensive investigation of groundwater flow to intercept sources may be futile.

4.2.5 Parallel treatment trains

If a unit requires maintenance, where will the flow go during this work? For some processes such as an aeration pond, maintenance might be done while flow continues. For example, precipitated iron can be dredged out under normal flow conditions. On the other hand, when a BCR has exhausted the organic or alkalinity source, the BCR volume must be replaced. In most water treatment systems, parallel units are included to allow for such maintenance. It is similarly preferable that BCR systems be built with parallel units and piping to allow any given unit to be taken offline as needed.

4.2.6 Full-scale systems

Upon completion of the field pilot test, full-scale design should take into consideration seasonal fluctuations in flow rate and in the chemical composition that may not have occurred during a pilot scale test if it was conducted for a short time. Fluctuations in loading can be managed using equalization ponds, flow bypass, recirculation or other pretreatments.

The field pilot-scale test provides data for scale-up. In some cases that scale-up is simply an application of the rates found at pilot-scale. For example, biology and MIW chemistry occur at microscale or smaller, therefore scale-up might be thought of as simply applying areal and volumetric rates from the smaller-scale tests to the total flow and load that require treatment. Scale-up, however, can both positively and negatively affect other issues.

On the positive side of increasing scale is the diminishing impact of edge effects. Edges promote heat transfer and also are frequent locations for short-circuiting. As the volume of a system increases, the fraction of volume exposed to surfaces decreases, therefore temperature variation due to ambient air contact declines and the fraction of edge-influenced, short-circuiting flow (along walls) also decreases. On the negative side, the availability of materials can be a problem, and construction of systems with unusual or rare materials at full-scale can be a challenge.

4.3 BCR Design Components

While BCRs are the focus of this design guidance, the information presented in this section applies not just to BCRs but also to related systems termed anaerobic wetlands, sulfate-reducing bioreactors, and permeable reactive barriers when these systems are designed to produce sulfide by the reduction of sulfate through biological means. This process is not the sole form of metals removal observed in BCRs. Metals are removed by precipitation as sulfides and adsorption to organic and inorganic ligands (including ion exchange), but in some cases also may be removed by precipitation as oxides or carbonates. In addition to metals removal, acidity may be neutralized both as sulfate reduction consumes protons and produces hydrogen sulfide and more importantly by carbonates, most often limestone, in the substrate that dissolves and provides alkalinity (see Section 1.3).

4.3.1 Flow direction

A BCR is unlikely to benefit from surface flow. Exposing the MIW to air as surface water will increase the oxygen load into the reactor, resulting in the consumption of organic matter to decrease DO rather than as a source of electron donation for SRB. Instead, the influent to a BCR is best distributed through a highly porous lens such as a layer of large gravel. The high hydraulic conductivity of gravel as compared to BCR substrate (10 cm/s compared to 0.001 cm/s) will give good distribution of the fluid to the substrate (although see discussion of influent distribution in Section 4.4.2.2), and the stagnant air in the pore space of the gravel layer greatly reduces oxygen transport rates from the air.

Although subsurface flow is preferable in a BCR to surface flow, the direction of flow, whether downflow, upflow, or horizontal flow (as shown in Figure 4-3) has not been shown to affect removal (Section 4.1). The biology and chemistry occurs at a scale where gravitational direction is unimportant. Hydraulics are, however, important and differ significantly between the flow directions. Vertical flow systems are generally downflow because an upflow system is more prone to plugging due to solids in the influent water. Compared to horizontal flow, vertical flow gives a greater allowance for head loss. In a horizontal flow system, excessive head loss will result in overland flow, bypassing treatment. On the other hand, the maintenance of horizontal flow systems may be easier than for vertical flow systems, because the location of a short circuit can be accessed. In a vertical flow system, localized failure may be hidden under the distribution layer.

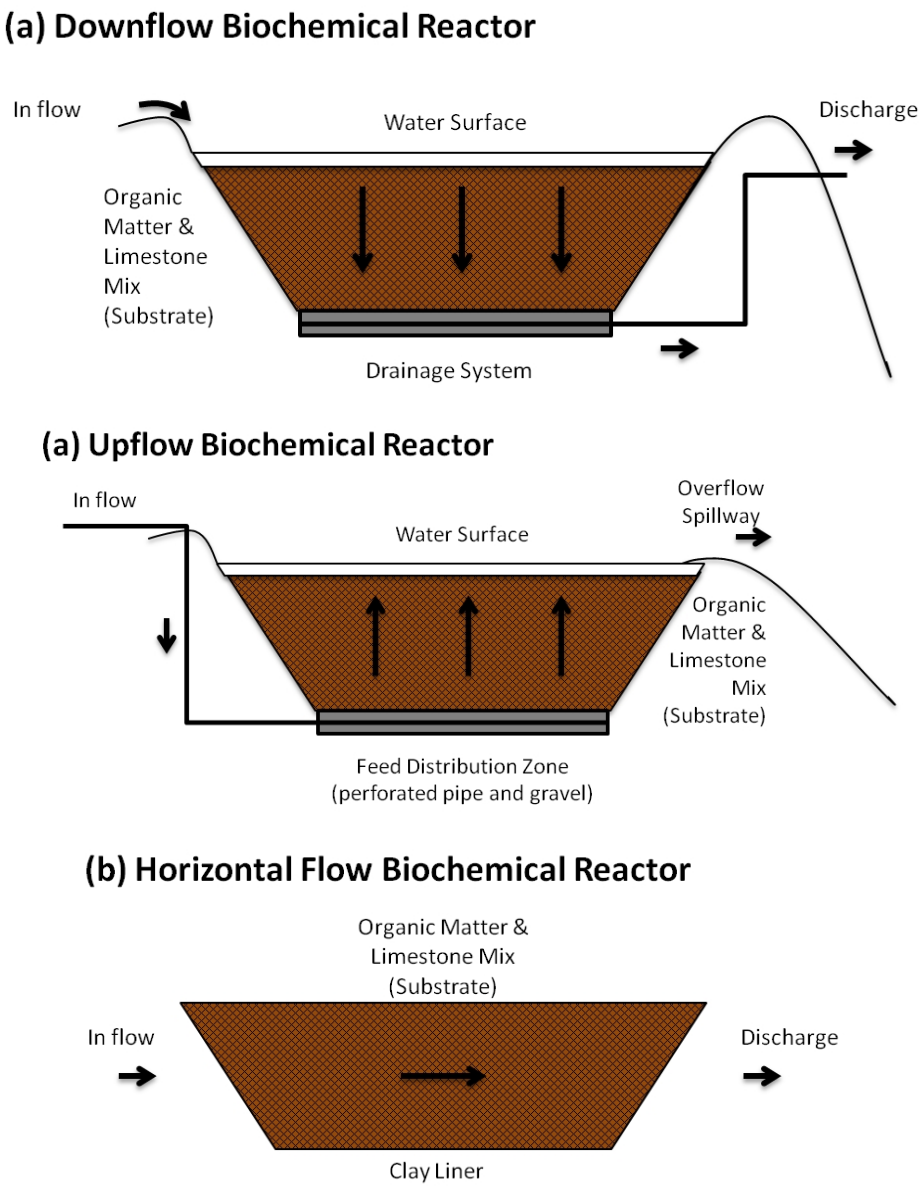


Figure 4-3. BCR bed configuration for (a) vertical flow and (b) horizontal flow.

Source: Dr. Mark Fitch, Missouri University of Science and Technology

Another consideration for flow direction is oxygen supply. Horizontal flow allows for oxygen to contact the media, and thus presumably more oxygen flux comes into the media. Added oxygen flux may shorten the lifespan of the reactor because the organic is more rapidly consumed, but may also result in more small carbon for use by SRBs and thus result in a higher rate of sulfate reduction. At present, no studies have compared the flow configurations to demonstrate whether organic degradation rates differ (Deitz and Stidinger 1996).

4.3.2 Media Design

Section 3.0 describes the substrate selection process and subsequent testing. The media used in a solid substrate BCR should be designed and evaluated with six objectives in mind (see Table 3-1):

- Long-term organic source. The media should provide sufficient BOD to both deplete any influent oxygen and to maintain a sufficient rate of sulfide generation.
- Short-term organic source. The media should have enough readily degradable organic material to allow the system to quickly become anaerobic and create sufficiently reduced conditions that favor SRBs.
- Organism source. The media should have seed organisms or be seeded with a known culture of organisms to establish anaerobic conditions, cellulose degradation, and sulfate reducers. Batch studies of inocula have shown that media from existing BCRs give a shorter period to produce sulfate reduction and the appearance of H_2S and a somewhat higher rate of sulfate reduction over several weeks than dairy manure (Pereyra et al. 2008). Often there are lagoons or slime dams at mine sites; the sediment from these sites can be used as an inoculum for the reactor (Mary Deflaun, Geosyntec personal communication). A combination of both aerobic and anaerobic inoculum works best since oxygen and nitrate in the influent must be consumed before the anaerobic processes commence. Often, a good inoculum for a full scale BCR is the media from a pilot scale unit operated at the same site.
- Alkalinity source. A BCR may require an alkalinity source because most MIW is acidic, and biochemistry cannot easily increase pH. Alkalinity addition may not be needed for neutral or alkaline MIW. However, additional alkalinity can suppress fermentation bacteria in short-term electron donor materials during start-up. This alkalinity may also protect the pH-sensitive bacterial community during short-lived acidity upset conditions.
- Bulk for hydraulic conductivity. If the bed is impermeable, no water will flow through it.
- Attachment sites for organisms. Most organisms live in small biofilm communities attached to surfaces, which are provided by the solids in the media.

Typically, 10-30% by weight limestone gravel is included in the substrate blend to provide a source of alkalinity and bulk. The long-term organic substrate of choice is often wood. The preferred wood particle size is currently an unresolved question. Smaller wood particles increase the rate of soluble organic evolution due to a greater surface area, while larger wood particles last longer and maintain lower concentrations of soluble organics in the BCR. Combinations of larger wood chips and smaller sawdust materials in a BCR may provide both these long and short-term benefits.

The concentration of soluble organics in BCRs is thus a balance of the competing needs of longevity (large pieces) and surface area (small pieces) relative to the rate of decomposition of the wood particles. Characterization of several BCR organic substrates was offered by Seyler et al. (2003). Most wood has very little rapidly degradable material for consuming influent oxygen and creating initial anaerobic conditions. Thus, wood-based BCR substrates may benefit from admixtures (Section 2.8 and Section 3.1) such as unusual solid-phase amendments or liquid substrates (e.g. chitin).

Chitin is a solid-phase substrate for use as a BCR media component is material made from crustacean shells, which contain chitin. Chitin is an aminated glucose polymer with small calcium carbonate particles. The crustacean shell has a chemical composition of approximately 20% chitin, 40% chitin-bound calcium carbonate, 30% protein, and less than 9% nitrogen (Venot et al. 2008). This material is a long-term source of organic substrate and provides alkalinity through calcium carbonate dissolution. Manure can also be added to a BCR to supply a culture of organisms for start-up and a short-term soluble organic source.

If there is an available source of liquid substrate, the system is accessible to refill the liquid substrate, and the system can be monitored regularly (weekly to monthly) for flow rate adjustments, then a liquid substrate reactor may be considered for long term sustainability, particularly if space is limited. In this type of BCR, the media might simply be gravel or local trap rock with an initial inoculum of organisms. A thin layer of manure is often used at the bottom of the BCR to provide the inoculum. The liquid organic feed rate should be adjusted according to flow and chemistry. A liquid base such as 25% sodium hydroxide can also be used to control pH and increase activity (Leviathan Case Study, B.8).

After the media has been tested at pilot-scale, the design of the BCR is relatively straightforward. If not done during the testing phase, the substrate materials should undergo comprehensive testing for incidental contaminants that could be introduced into the system. The pilot-scale unit allows determination of a sulfate reduction rate, and this value is then used to find the total media volume required (see Section 4.4.2). Where a sulfate reduction rate can be derived or assumed from similar sites a sulfate reduction rate may not be derived through testing. in this case the hydraulic Residence time will assume a default Sulfate reduction rated and the BCR designed accordingly (see Section 4.4.2).

Each unit of the passive treatment system is designed for an expected load of water, acid, and metal. The influent may be subject to high variability in load, particularly on a seasonal basis such as due to snowmelt. Design the influent capture system to provide loading control for variations from the expected loads. Excessive loading can affect different units:

- Excessive acidity can overwhelm alkalinity production and as a result inhibit sulfide production in a BCR. SRB are generally sensitive to decreased pH. The system described in Section B.9 is one such example.
- Excessive flow can scour solids from any unit, but is particularly troublesome when it occurs in sedimentation ponds and APC units.
- Excessive water velocity through biological processes can cause washout, which is the loss of biomass. The result is a steep decline or total loss of activity.
- Excessive flow also can promote short-circuiting in BCRs. The increased head pushes more flow through an area of higher hydraulic conductivity and the resulting higher velocity scours more material, thus decreasing conductivity further until a channel is formed.
- Excessive iron or aluminum can escape aerobic conditions and instead oxidize and precipitate in the anaerobic zone or compete for sulfide and thus decrease efficiency for other metals.

Loading control ideally means that a storage system is included to allow processing of all the influent MIW or that the system was designed in anticipation of variable loading. In the event of extreme variability, however, it might be necessary to (a) only partially treat all of the MIW water, such as processing through an ALD pretreatment but bypassing some of the MIW around the BCR, or (b) treat only a part of the water and bypass the remainder past the entire sequential treatment system. Mixing of bypass water with treated effluent from the BCR may be considered. The designer must analyze the required cost and construction of storage capacity compared to increased treatment capacity. Bypass of water is generally a poor option, since protecting the receiving body of water is the goal of the system and in some cases untreated bypass water may not meet expected permit conditions

If the increase in load is due not to increased flow but rather to changes in concentrations, then the system should be designed for that higher concentration. The system is thus over-designed for typical values but is able to treat the maximum. Otherwise an influent control system with instrumentation and a programmable logic controller (PLC) could be used to shift flows. For example, a decrease in pH could result in a PLC opening a valve for additional alkalinity input through an on-demand active system.

4.3.3 BCR Sizing and Media Depth

The volume of a BCR and the quantity of the media within a BCR is established by considering the chemical reaction kinetics and hydraulics conditions that are expected. Reaction kinetics that may be considered include: a sulfide generation rate, the rate of removal of metals and/or the yield of dissolved organic carbon from cellulose degradation. Typically the slowest chemical reactions control the BCR size. Hydraulic conditions that are expected should consider the nominal and peak flows and the duration and frequency of peak flows. Reaction kinetics and hydraulic conditions are typically compared by calculating the residence time, or empty bed contact time, within the BCR. Professional judgment and experience are then used to establish a safety factor on the minimum design volume and media quantity established from analysis of reaction kinetics and hydraulic conditions.

An example of how to size a BCR using the observed sulfide production rate and influent metals loading to the BCR is provided in Appendix C (Example 4-1). An allowance, or safety factor, for media aging is included that accounts for reduced sulfide production rate over time and design flow rate. Although the literature reports a range of sulfate reduction rates, generally the range of 100 – 300 mmol/m3-d has typically been used for pilot BCR design.

It is important to account for the dynamic BCR behavior over time. The rate of decomposition of organic matter within the BCR media changes over time leading to changes in the microbial activity within the BCR over time. For example, the rate of sulfate reduction (sulfide production) will typically increase over time during BCR start-up. As the microbial population density within the BCR zones (oxic, transition and sulfate reducing) increases the rate of production of bioavailable organic carbon and sulfide often increases. As the readily degradable fraction of the media is

depleted and microbial cellulose degradation provided a larger fraction of bioavailable carbon (electron donor) for sulfate reduction and sulfide production.

Pulles et al. (2001) showed that in the early operational days of a bioreactor there is a large loss of both microbes and organic carbon as it is flushed from the system. This was also noted by Glombitza (2001) and others. This initial period of high cell yield and loss of organic carbon will be followed by a period of slower kinetics driven by use of cellulosic material that is eventually exhausted. Possible approaches to address these issues include recirculating BCR effluent during start-up and providing supplemental carbon if sulfate reduction rates decrease below design levels.

The design approach presented above assumes that metal removal is the primary purpose of the BCR and that complete removal of sulfate or nutrients is not a design objective. In the case that sulfate or nutrient removal is required to meet discharge requirements, the residence time of the BCT may by driven by reactions beyond sulfide production. Additional design considerations may results such as removal of excess sulfide or methane. Direct scaling of sulfide production rates beyond the range of 100 – 300 mmol/m^3-d may be inappropriate because in general the rate of biological processes decrease in rate as the concentration of the limiting reactant decreases (bioavailable carbon or sulfate). In this, twice the BCR volume may not double the removal of a particular constituent of interest.

Similarly, the designed is cautioned against changing the depth of a full-scale BCR as compared to a pilot system. The maximum media depth is a function of the compressibility of the media. Under totally saturated conditions media compression may be mitigated and become negligible due to material buoyancy. For some media, compression may reduce the hydraulic conductivity of the bed when the bed depth is greater than one meter. Media bed depths greater than a meter should be evaluated using geotechnical principals. A maximum media bed depth of roughly 2 meters is advised unless the media is dominated by inorganic, large particles. For example, a limestone bed is expected to have minimal compression as compared to a manure/straw bed.

The hydraulic design of a BCR is illustrated in Example 4-2, Appendix C. Since the depth of a solid substrate BCR is limited by compressibility of organic media, the surface area of the BCR increases with increasing flow rate. In some cases, the peak seasonal flow may lead to lower hydraulic residence time than required for complete treatment of all MIW constituents and nutrients. The maximum surface area of the BCR may be limited by the area of land area available at the site. In some cases, this may lead to selection of a liquid substrate BCR over a solid substrate BCR or a seasonal hybrid design. Example 4-2 in Appendix C illustrates linkage of hydraulic calculations and media volume estimates.

4.3.4 Sampling Points Within a BCR Substrate

Sampling ports could be incorporated into the design to allow for monitoring conditions along the depth (flow path) of the BCR (see Section 6.3.1). Sampling ports could be provided by embedding plastic pipe of appropriate size in various locations and at various recorded depths within the BCR bed. An example of such a sampler is shown in Figure 4-4. In some cases, socks of media

contained in plastic net are placed in samplers to allow for withdrawal of substrate samples as the media ages and assay of organic depletion rates.

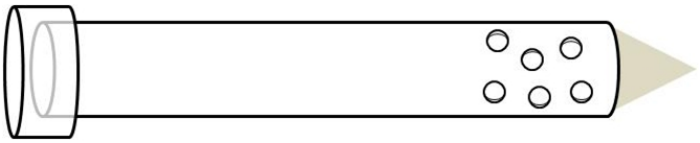


Figure 4-4. Sampling pipe/piezometer. Note perforations to provide pore water samples, removable cap, and wooden cone for driving into substrate.

Source: Dr. Mark Fitch, Missouri University of Science and Technology.

Sampling points should not be placed near one another because the pipe may act as a surface to allow for some short circuiting and because placing substrate between co-located pipes may result in voids. Although samplers with conical ends as shown in Figure 4-4 have been driven into BCRs after construction, this practice likely causes the opposite problem: compaction around and beneath the sampler.

4.3.5 Aerobic Polishing Cell

Many BCR installations include an APC as a final element. An APC is a shallow pond that aerates the treated water, allows some degradation of organic matter escaping the BCR, and also retains suspended solids (see Section 2.5). If manganese removal is a treatment goal, a limestone or other carbonate rock bed can be included in the APC (see A.1.4 Manganese); however, the resulting pH may be 8 or greater. Often APCs are shallow enough that plant growth will occur and thus these APCs can also be considered aerobic wetlands. Plant uptake may help to remove trace metals. In conjunction with proper sedimentation processes, the APC can produce a final effluent water quality that is acceptable for discharge to receiving streams or other water bodies.

Butler et al. (2011) tested BCR effluent from four different systems for aquatic toxicity using USEPA's whole effluent toxicity (WET) tests. The effluent was toxic, but further aeration removed toxicity for some samples. Aeration of the effluent removes dissolved sulfides and increases the DO content prior to ultimate discharge. If the BCR discharges over a weir with a few feet of drop, such aeration and concomitant toxicity reduction is likely to result. If WET testing standards must be met, additional bench testing may be necessary.

4.3.6 Outfall

The outfall is usually the point of compliance at which water quality monitoring is required as discussed in Section 6.2. Depending upon the regulatory discussion and expected operating permit, an outlet monitoring structure such as a small, automated sampler and some instrumentation might be required.

4.4 Hydraulics

The application of hydraulics and hydrology is a crucial element of BCR design. Water flows downhill, and the design of the system must anticipate that sometimes downhill in terms of the energy grade line is not literally downhill. Consider these three major aspects of hydraulics in system design:

- The energy and hydraulic grade line (HGL) should be calculated for the entire system (BCR and pre- and posttreatment processes), and in so doing each unit of the system must be considered in terms of head loss. This approach should include an initial rough estimate and a later check of calculated values when units have been sized, with production of a process flow drawing including the energy grade line (EGL) and HGL.
- The BCR bed is a heterogeneous porous media, which changes with time due to deposition, filtration, scour, development of biofilms, and chemical reactions. The flow path within the bed, should be calculated and allow for both changes in hydraulic conductivity over time within the bed and changes in the amount of short-circuiting.
- The system is subject to variation in influent flow rate, such as from precipitation events or snow melt. Provisions should be made for handling stormwater and snow melt flows, which are likely as both sheet flow into the BCR and variations in MIW flow rates.

4.4.1 Sequential Treatment Hydraulics

Early in the design process, the hydraulic profile of the discharged and treated MIW should be estimated. Presumably the MIW emerges at the surface at some elevation, and the receiving body of water is at a lower elevation. Flow through a BCR and any associated pre- or posttreatment will result in head loss. The head losses for each element of the system should be estimated, including piping and flow control and measurement structures. Estimated head loss from the source to the receiving body of water can be plotted as shown in Figure 4-5.

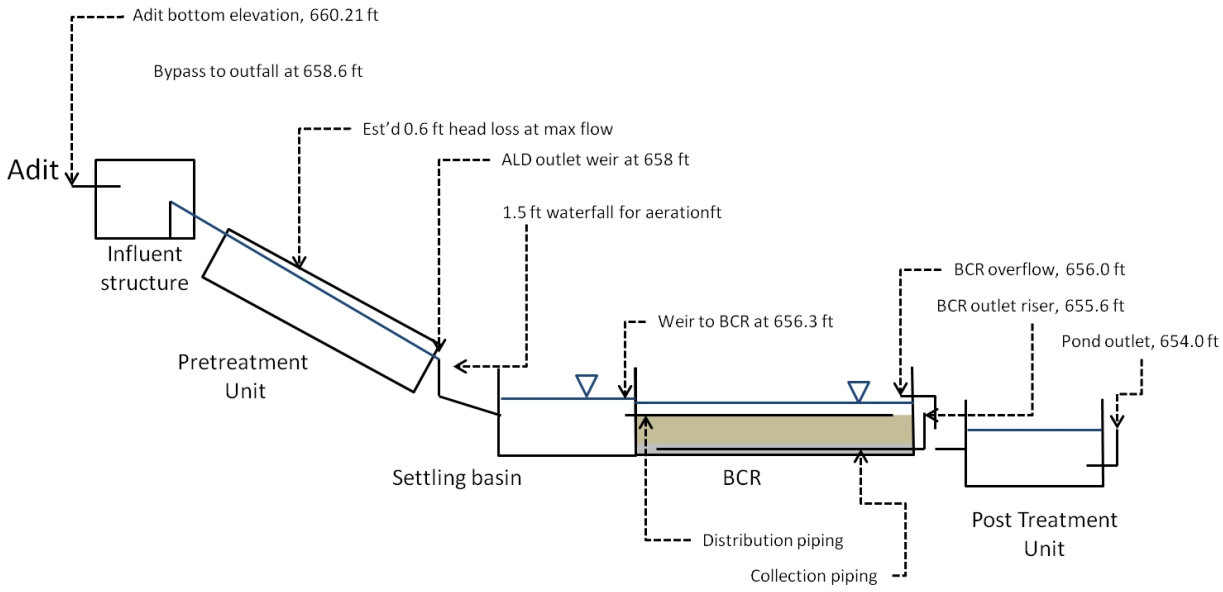


Figure 4-5. Example of initial system hydraulic profile.

Source: Dr. Mark Fitch, Missouri University of Science and Technology.

The purpose of this initial profile is to show whether there is a clear hydraulic issue requiring a pump to increase head or, in the case of excess energy, dissipation of energy to prevent erosion. This first analysis is not a detailed engineering design, but rather a back-of-envelope estimate.

After the various units have been designed, perform a detailed analysis of friction losses, and plot the EGL and HGL against the flow pathway. Determine or check sidewall depths of open units from these grade lines. Compare overflow levels to the expected water depth. In most cases the expected weir depth of the BCR outflow structure is determined from the calculated head loss through the BCR subtracted from the EGL at the BCR inlet.

In determining EGL and HGL, consider not just the maximal value flows, but also the average and minimum expected flow rates. The system should be designed to accommodate the highest flow, but generally run at an average flow. In some cases, the BCR freeboard is designed to allow for storage of excess water, where outlet flow is determined for the bed at the expected water level and accumulation is the difference between inlet and outlet flows during the time of excess flow multiplied by that time. The rising water level (head) results in increased effluent flow. As the head increases and therefore flow increases, the metals and acid load to the bed also increases and the velocity in the bed is more likely to scour out organisms and deposited metals. Both increased loading and scour can result in BCR failure. At minimum flow rates, undesired deposition might occur in piping and channels, and the head losses through most beds decrease, which might require adjustment to the height of weirs to prevent water level decline exposing portions of reactors to oxygen.

4.4.2 Unit Hydraulics

The BCR and associated units may be thought of as having four possible governing hydraulic processes:

1. Head loss due to flow through porous media, usually characterized by Darcy's law. In addition to the BCR unit, examples are ALDs, open limestone drains (OLDs), RAPs, SAPs, and wetlands.
2. Open basins in which the water hydraulic grade line will be equal to the elevation of the outlet weir or other control structure. Examples are aerobic wetlands, settling basins/clarifiers/sedimentation ponds, individual sub-basins within a cascade system, and aeration ponds.
3. Head loss due to friction in pipes and channels, usually determined by Darcy-Weisbach, Hazen-Williams, or the Manning equations. Note that OLDs and ALDs appear to be channels but should not be analyzed as simple open channel flow due to the porous fill.
4. Energy gain from mechanical input. Generally mechanized input is from pumps, but can also potentially come from pressurized air such as in jet aeration.

The decrease in head along the path of flow, dh/dL, is the hydraulic gradient in the direction of flow. For the entire system, this value integrates to the head loss, Δh, divided by the flow length, L. Most often the hydraulic gradient is approximated as this system difference, $dh/dL = \Delta h/L$. If the hydraulic conductivity of the homogeneous porous material is known, the flow rate may be calculated, or for a desired flow rate, the necessary head can be determined.

Although the term permeability is sometimes used interchangeably with hydraulic conductivity, the two are related as:

$$K = k\, g\, \rho/\mu$$

where:

k = permeability, m^2

g = gravitational acceleration, 9.8 m/s^2

ρ = density of the fluid, which for water at room temperature is approximately 1000 kg/m^3

μ = dynamic viscosity of the fluid, which for water at room temperature is approximately 0.009 P = 0.009 g/cm/s = 0.0009 kg/m/s

Measuring permeability in an operating BCR needs to be approached with an understanding that the value observed can be biased by several factors, the most important of which is the tendency of substrate to generate various gases such as carbon dioxide and hydrogen sulfide. True permeability or "saturated hydraulic conductivity" is predicated on a fully saturated test sample. The likely presence of gas bubbles in BCR substrate makes this measurement virtually impossible, so any per-

meability values derived from the static head elevation difference between the standing water on top of the cell and the outfall should be termed "apparent permeability".

To be even more precise in this calculation of apparent permeability of substrate,consider the following:

- the friction losses in the pipes and fittings and the gravel bed (i.e., the drainage layer) that is typically placed on the floor of the BCR, beneath the substrate
- heterogeneity of the substrate itself
- the tendency of evolved gas bubbles to create preferential vertical pathways as the bubbles migrate to the surface of the substrate
- potential "dead spots" around the perimeter of the BCR
- seasonal variations in flow rate, temperature, and associated microbial activity that could affect the kinetics of gas evolution

Thus, even measuring "apparent" BCR substrate permeability should be considered an inexact science and values derived need to be considered in the context of the operating conditions.

The hydraulic loading, Q/A, is also the flux, q, and may be termed the Darcy velocity; for a downflow BCR, this is the surface loading applied to the BCR, where Q is the flow and A is the area footprint of the BCR. The Darcy velocity is also the apparent velocity of water through the porous material ignoring that the material is porous. The actual velocity of the water in the pores is

$$v = q/\eta$$

where:

v = velocity, m/s

q = flux or Darcy velocity = Q/A, m/s

η= porosity, dimensionless

For a BCR, apply the above equations generally to determine the relationship between flow and head. Darcy's law can be used to determine head loss through the substrate, flow rate through the substrate, and residence time in the substrate. Head loss is simply Δh as determined for the given substrate hydraulic conductivity (K) and length (L) at the specified flow rate (Q) through the area of flow (A). Similarly, the flow rate can be found for a given applied or constant head. The hydraulic residence time (HRT) is slightly more complex. Hydraulic residence time is the mean length of time a fluid element is in the volume of interest (the BCR), and often is defined as:

$$HRT = V\eta /Q (= AL\eta/Q = L\ \eta/q = L/v)$$

where:

HRT = hydraulic residence time, s

Because estimating the porosity of a system is difficult, in many cases the empty bed contact time (EBCT) is used instead of the hydraulic residence time.

$$EBCT = V/Q \ (= AL/Q = L/q)$$

where:

V = volume

In addition to accounting for head losses, certain hydraulic structures should be considered:

- influent capture
- influent distribution
- weirs
- overflow structures and loading control
- pipes and cleanouts

Each of these structures is discussed briefly in the following sections.

4.4.2.1 Influent Capture

MIW may emerge from a seep, a borehole, an adit, or some other source. In cases in which the MIW emerges at the surface with no existing structure intercepting the flow, design a structure such as a spring box or cased pipe to capture that flow. The influent structure acts as a funnel to bring the water to a conduit of some sort. This structure is usually installed prior to pilot-scale testing in order to provide a location at which flow rate is measured. In some cases, the MIW flows from an existing structure or channel, in which case connection of the system to that structure must be designed. Factors to consider in this design include the geology and topography at the site, the anticipated or known flow rate and its variability, and the chemistry (potential for corrosion, deposition, and redox state) of the water.

A common approach for influent capture is to add a manhole or similar structure that allows overflow along the existing channel or a new pipe and diversion to the new passive treatment system. An example is shown in Figure 4-6.

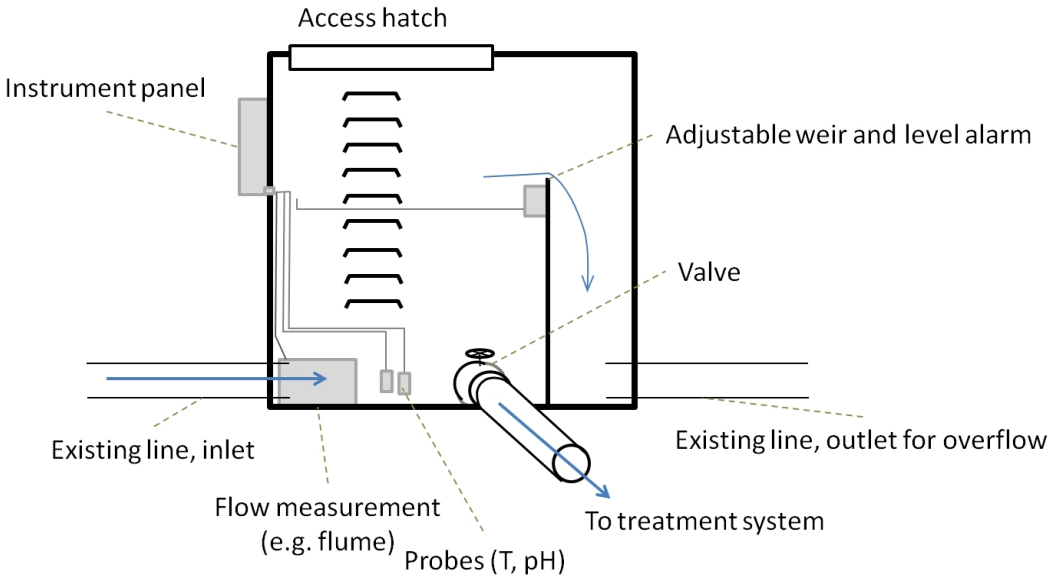


Figure 4-6. Example of an influent structure for an existing line.

Source: Dr. Mark Fitch, Missouri University of Science and Technology.

Provisions for overflow and bypass of the passive treatment system should be included. As an example, the adjustable weir shown in Figure 4-6, allows excess water to overflow and provides a bypass when the valve to the treatment system is closed. Potential for plugging of the influent capture piping should also be considered in design (for example, provide access for maintenance via cleanout ports).

4.4.2.2 Influent Distribution and Effluent Collection in BCRs

Distribution and collection appear simple: influent is piped into the BCR and distributed through a series of pipes or as free water; treated water is collected through a system of perforated pipe in a gravel bed. The design of these systems, however, is more complex than some designers anticipate. Issues can include (a) failure to analyze edges as separate subsections, (b) clogging of pipes due to precipitate formation and (c) failure to account for gravel and piping losses.

As mentioned above, a source of problems in some BCR systems has been the failure to calculate head losses or flow rates in subsections at the edge of the BCR. For example, consider the sloping side to a downflow BCR where flow is collected by a layer of gravel and sand. If the thinning bed of substrate is underlain by the collection system, flow preferentially will go through this shorter area of the bed. On the other hand, if there is no collection system under the thinning bed of substrate, this volume at the edge has a lower flow rate than the average for the system because the path length to collection is longer. If the areal load is calculated based on the footprint, errors occur because a portion of that surface area has a lower flow rate than average and the remaining area has a higher flow rate than expected. In horizontal flow systems, these edges have been used for flow distribution. Thus if the BCR design is a non-uniform shape, as is often the case, differing sections should be accounted for and analyzed as discrete elements.

Another source of potential error is to ignore the impact of gravel and piping on flow. Consider a vertical flow system in which parallel perforated pipes separated by four feet on center deliver water to the gravel layer of the BCR. The water flowing down the pipes loses a small amount of head by friction, so a fraction more water is likely to be delivered at the inlet of the system. It might seem that the head loss through the holes in the pipes is small, but flow direction changes at the perforation orifice, and the head loss is not well-described by a single simple coefficient in the Darcy-Weisbach approach (Clemo 2006).

Danehy et al. (2002) claim that flow through the perforations is large enough that a standard 4-inch perforated pipe reaches full flow (for collection) or discharge (when distributing MIW) in about ten feet. Long lengths of perforated pipe therefore may not function as expected. After exiting the perforations, the water moves to the BCR substrate through the gravel, resulting in a flow distance of up to two feet. Unless the design engineer, based on testing, requires geotextile installation, only use geotextile to protect geomembrane integrity. If a geotextile is used to separate the gravel and substrate, preventing fines from clogging pore space in gravel, the geotextile becomes an additional component of head loss. Note that (a) such geotextiles should not be installed atop substrate due to their potential to trap gases, and (b) the geotextile might instead clog with fines. These combined losses may be mere millimeters of water compared to the meter of BCR substrate, but as a result the flow rate at the location furthest from the inlet could be a few percent lower than at the inlet assuming that the gravel and the BCR media are homogeneous in terms of hydraulic conductivity.

Based on these constraints, consider the flow and head loss in any passive treatment unit as occurring through the variety of flow paths rather than assume the system flow path is an average whole (see Figure 4-5). Design the influent and effluent distribution systems to give equivalent frictional losses across the entire flow area of the unit regardless of distance from the influent or outlet point. At the system edges, though, it is better to not deliver as much flow since the impermeable wall is a frequent partner in short circuit development. One example of such wall short-circuit was observed at a site in Vermont, where upon freezing water expansion pushed apart the media and the wall, with a visible layer of ice between the substrate and the reactor (Smart et al. 2008).

The dominant head loss is due to the hydraulic conductivity of the media, which generally is lower than the influent and effluent systems. Few published reports exist for values for K in BCR materials. In Table 14-1, Bolis et al. (1992), hydraulic conductivity of initially dry substrate is shown to fluctuate significantly, and soaked material may have values lower by an order of magnitude than seen after a few months. A similar increase in K was seen in an unpublished study of hydraulic conductivity (Ross 1997).

Table 4-1. Hydraulic conductivity of BCR substrates

Substrate	Range of K (cm/s)	Reference
77% limestone (1/8-1/2 inch), 14% alfalfa, 9% inoculum from another BCR	0.001 – 0.012	Wildeman et al. 1992
75% cow manure, 25% potting soil	$1´10^{-4}$ – $6´10^{-3}$	Wildeman et al. 1992
75% cow manure, 25% top soil	L.T. $5´10^{-5}$ – 0.005	Wildeman et al. 1992
5% peat moss, 5% alfalfa hay, 20% chert	$1´10^{-4}$ – $7´10^{-4}$	Ross 1997

Table 4-1. Hydraulic conductivity of BCR substrates

Substrate	Range of K (cm/s)	Reference
gravel, 3% aged manure, 2% biosolids, 15% sand, 50% chip bark		
Ibid, limestone instead of chert gravel, compacted to half initial dry volume	$3´10^{-6} – 4´10^{-4}$	Ross 1997
Chitin	Based on the blend of material containing Chitin	JRW Personal communication
Chitorem	Based on the blend of material containing Chitorem	JRW Personal communication
Mushroom compost	$3´10^{-4} – 0.014$	Lemke 1989 as reported in Wildeman 1992
100% organic compost[1] (biosolids and wood chips)	$7.8´10^{-4}$	Mueller et al.1996
70% organic compost, 30% gravel	$7.4´10^{-4}$	Mueller et al.1996
40% organic compost, 60% gravel	0.032	Mueller et al.1996
100% gravel	1	Mueller et al.1996
100% gravel	> 0.6, > 1.02	Mueller et al.1997
20% organic compost, 80% gravel	0.8	Mueller et al.1997
50% organic compost, 50% gravel	$< 5´10^{-4}$	Mueller et al.1997
Note 1: The term "compost" implies a product of the composting process; many early researchers may not have made that distinction and labeled any mixture exhibiting compost-like organic properties as compost (Mueller et al. 1996).		

Unfortunately, real materials like soil or BCR substrate are non-homogenous and have widely differing properties. For hydraulic conductivity, values may be different by more than an order of magnitude depending on the direction of measurement, and compaction (during construction) may decrease hydraulic conductivity by two or more orders of magnitude. As an example, Figure 4-7 shows the change in hydraulic conductivity of a freshly mixed BCR substrate (50% chip bark, 15% sand, 20% limestone gravel, remainder moss, hay, straw and manure) over several weeks as water flowed through the substrate. Bolis et al. (1992) reported that hydraulic conductivity of initially dry substrate fluctuates significantly, and soaked material may initially have values lower by an order of magnitude than the hydraulic conductivity seen after a few months. A similar increase in K after an initial decline was seen in an unpublished study of hydraulic conductivity (Ross 1997).

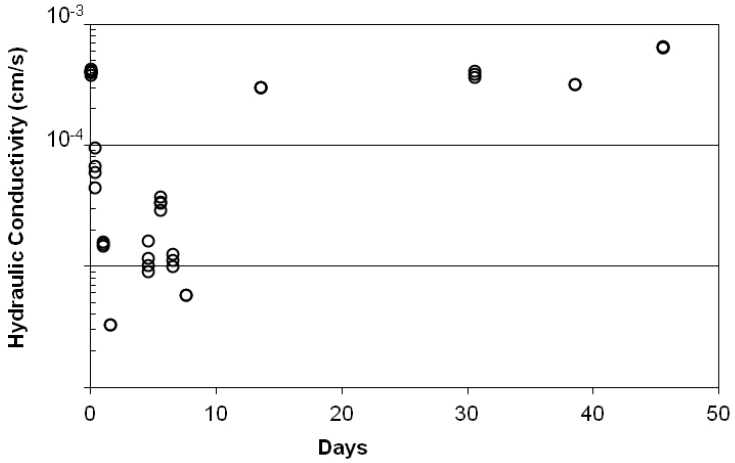


Figure 4-7. Variation in hydraulic conductivity.

Source: Ross 1997, modified by Dr. Mark Fitch, Missouri University of Science and Technology.

If one portion of the substrate has a higher hydraulic conductivity than others, more water will flow through this location. The higher hydraulic conductivity is probably due to larger pores related to local structure and mixing of the original substrate. The flow of more water at this location increases the shear and also delivers more mass of chemical, which in turn increases the observable rates of reaction. In BCR substrate, this flow increase results in more particle scour, enlargement of pores, more rapid depletion of organic matter, and therefore some further increase in porosity. Thus the porosity, already larger than average, is increased, and because pores are larger, hydraulic conductivity increases and even more water flows at the location. At an extreme, all substrate would be cleared and the location essentially becomes a pipe, transporting water through the substrate without treatment. This phenomenon is referred to as “short-circuiting" (Section 4.4.5).

An illustration of flow hydraulic calculation is provided in Appendix C, Example 4-2.

4.4.2.3 Weirs

The water level in many units is controlled by a weir at the outlet. Because many of these treatment units have an uncertain head loss due to flow through porous material, t this outlet weir should be adjustable in height. Several configurations are possible for adjustable weirs (Section 6.3.2.1). Although weir gates are common in the water industry, frequent adjustment is not generally needed in a passive treatment system, and the components of weir gates generally need routine exercise to prevent seizing. Weir gates are therefore not usually a good choice for a BCR system.

Weirs using a sleeve with inserted blocks or wood boards are commonly used in BCRs. Wood boards have certain drawbacks (including a tendency to leak at joints) and are difficult to install level and at the precise desired elevation. Rubber-coated metal may be a better weir board for BCRs.

A frequent choice of adjustable weir is a vertical polyvinyl chloride (PVC) riser. A simple pipe can be cut to the desired elevation, or if too low, may be raised by addition of a union. Another option is to use a flexible rubber coupling as a reducing union for a small pipe, which can be adjusted vertically by loosening the coupling hose clamp and sliding the pipe up or down.

In addition to considering adjustable level control, also consider freezing in weir design. In many cases, the site will be cold enough that ice can form on exposed surfaces. For a weir, ice can raise the effective weir level. Avoid failure due to icing by placing the outlet structure in the berm or within an insulated or even heated structure.

One rare but significant problem with weirs is that the waterfall noise is an attraction for beavers. Minimize noise by designing short fall distances onto cobbles or enclosing weirs (for example, Clemson Pond leveler, www.clemson.edu/psapublishing/pages/afw/afw1.pdf).

4.4.2.4 Overflow Structures and Loading Control

In the case of excess influent to any unit or plugging of a bed, the design of the unit should include an allowance for overflow. That overflow might be channeled to the next treatment unit in a sequence or might go to a stormwater bypass. In many cases the overflow level is set by having a pipe run through the containing lined berm. If the pipe does not continue to a channel or additional piping, the resulting erosion scour effects must be considered.

If the influent flow rate to a BCR increases unexpectedly, the hydraulic loading to the BCR also increases. Increased load may result in failure, so the overflow level should be set for the maximum designed loading rate. This practice is illustrated in Figure 4-8. The BCR is designed for the expected load and the head loss through the substrate bed at that loading is estimated from the hydraulic conductivity observed in pilot testing. The outlet weir is thus set relative to the anticipated inlet or surface water level and overflow is included in the design to prevent loading failure. Estimate the maximum load allowable from the capacity to generate sulfide, the available sidewall depth, or the oxygen supply rate at which anaerobic conditions are inhibited. Then use this maximum load to calculate the expected head loss at which overflow should occur.

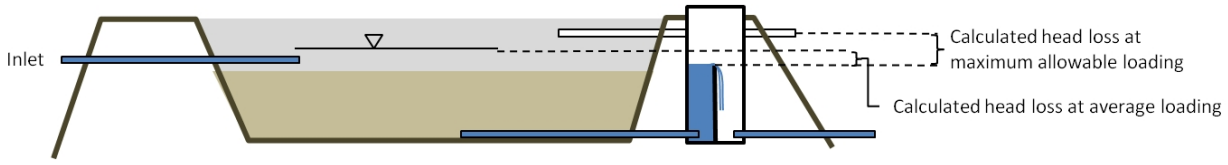


Figure 4-8. Weir and overflow level for a BCR.

Source: Dr. Mark Fitch, Missouri University of Science and Technology.

Figure 4-9 illustrates determining overflow level and shows a separate overflow line from the BCR outlet. The overflow can also connect to the outlet control structure so that treated and overflow water both exit through a single pipe. The overflow also can be set using an adjustable weir, allowing for changes in the field as operating experience is developed.

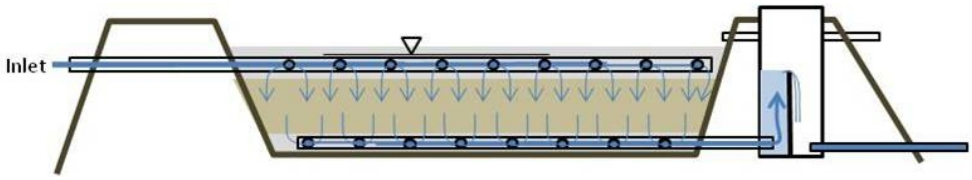


Figure 4-9. Section view of influent-effluent control systems.

Source: Dr. Mark Fitch, Missouri University of Science and Technology.

Screens should be included in the outlet piping to prevent entry of rats or larger animals. When accessing such a structure, be aware that snakes, spiders, and wasps find these structures to be excellent homesteads. Be alert for these animals and take appropriate precautions. Also be aware of the potential for H_2S gas in confined spaces.

4.4.2.5 Pipes and Cleanouts

The iron in most MIW, when oxidized, forms a dense solid capable of plugging pipes. Other metals, which form sulfides and carbonates, have the same potential to cause fouling. Deposition of solids causing this clogging in pipes can be minimized by maintaining high velocities in the range of 1 m/sec, but such high velocities result in high head losses. A different approach is to allow for deposition by increasing pipe size. Larger diameter pipes extend the duration required between cleaning and initially also minimize head losses in piping, although at a higher capital cost for larger pipes. A more general practice is to avoid the use of pipes when practical and instead use channels or subsurface drains. When pipes are used, minimize pipe run distances

When using piping, cleanouts should be used in piping systems to maintain and control deposition. The cleanouts allow access to piping in order to clean that pipe, and generally are installed at locations where deposits are likely to occur, such as at a tee location instead of an elbow or at the base of a pipe run. Because the BCR generates hydrogen sulfide, cleanout caps may accumulate gas and therefore require breather holes. This condition often occurs in cleanouts used for collection piping in vertical flow BCRs. In addition to simple mechanical access, cleanouts may be provided as a location to inject pressurized water or air to cause scour, or to inject and receive cleaning solutions. In general, pressurized air should be avoided in BCR systems due to the potential for pores to become vapor locked. A more mechanical cleaning option can be to run brushes down pipes or even to use a pipeline inspection gauge (PIG). When using a PIG, the system must be designed to include insertion (launch) and collection (receiver) points along relatively straight runs.

Pipe design must also consider placement. Because plastic pipes are affected by UV and piping on the surface will result in more variable temperatures, piping should be located underground when possible or surface piping should be adequately anchored to prevent movement of pipe due to expansion and contraction. In many locations the frost line depth might be considered sufficient insulation to prevent temperature changes. For underground installation, also consider the mechanical load that may be placed on a pipe. Specification of standards such as American Society of Testing and Materials (ASTM) D2665-11 (for PVC) and C2665-11 (for concrete) may be sufficient, but the designer should also consider the possibility of significant dead and live loads from

heavy equipment. Small-diameter pipes meet these specifications readily, but larger diameter piping may require thicker backfill over the pipe.

A final aspect of piping design to consider is material choice. Plastic piping is likely to be used in passive systems because most polymers resist acid (with some nylons and polyurethanes as notable exceptions). Not all plastics are the same in terms of chemical resistance, strength, durability, and other properties. For example, in some early BCR designs, high density polyethylene (HDPE) piping resulted in problems because of a tendency to float due to the low density of polyethylene. The relatively large coefficient of thermal expansion can also be a problem for long HDPE runs.

4.4.3 Stormwater

Because passive treatment systems for MIW largely consist of basins open to precipitation l and often are located in fairly steep terrain, the potential effects of precipitation should be included in the design. Rain and snowmelt is clean compared to the MIW influent, and should be excluded from entering the treatment system if possible to reduce the volume of water to be treated. Runoff control structures (i.e., stormwater diversions) should be built around treatment units (and on top of if covers are used) to divert precipitation from entering the system. In extreme some cases, the design can include construction of stormwater culverts to drain the site. Use a "design storm" for the area, such as the precipitation frequency data maintained by NOAA or the USDA Natural Resources Conservation Service.

Precipitation entering the open basins may affect the residence time in a unit. For example, suppose a BCR is designed to have a hydraulic load of 0.50 m/d. A 2-year 24-hour rain event might add 5 cm of water to the BCR basin. Assuming flow is controlled by the outlet structure, the loading rate during this 24-hour period increased to 0.55 m/d, and the residence time is about 9% less than expected.

However, rain may also affect the flow rate of the influent MIW. If the mine transports shallow groundwater, the rain event may strongly affect the flow rate of the influent. This additional flow is part of the reason that the design team should collect extensive flow data during the testing and data collection phase of the design.

4.4.4 Bed Clogging

Red mud can quickly foul a porous bed and cause hydraulic failure of a BCR. Although the term "red mud" is generally used in terms of alumina production for a strongly alkaline waste, some of the same components, iron oxyhydroxides and aluminum hydroxides, will form from many MIW sources. These solids form dense particles, which easily settle onto and are intercepted in porous beds, eventually plugging that bed.

Most BCRs are not currently designed for backwashing to remove plugging. The best method to control such clogging of the bed is to either prevent oxide formation or to pretreat the water to remove the oxides. In the former case, the MIW is protected from contact with the atmosphere and

minimal alkalinity is introduced. In the latter case, alkalinity may be introduced and aeration is allowed to occur (see Section 2.5 for further information).

4.4.5 Short Circuiting

Early prototype BCRs typically failed after two to five years of operation due to loss of uniform permeability, which led to short circuits, preferential pathways for flow, or plugging of the reactor substrate. Short-circuiting was not the sole reason for some BCR failures, however. Although short-circuiting is likely to occur, certain elements of design can help to minimize this phenomenon. These elements include the shape relative to the flow direction and the homogeneity of the media. Prevention of short circuit or flow through the reactor is favored in systems that are significantly longer (flow pathway) than they are wide. This size aspect is easier to achieve in a vertical flow rather than in a horizontal system. The media should be well-mixed, since variation in material sizes across a media bed will result in variation in hydraulic conductivity, which in turn means decreased residence time through those areas with higher hydraulic conductivity.

High loading rates tend to exacerbate short-circuiting. High hydraulic loading results in increased velocities in existing pore spaces, and thus increased scour of the pore walls, which can then increase the pore size. In addition to scour effects, high contaminant loading can affect short-circuiting. As reported in Section 4.4.2.2, if one portion of the substrate has a higher hydraulic conductivity that others, more water will flow through this location thus increasing shear and reaction with the substrate, potentially clearing (and scouring) pores and creating a piping affect. Tortuous, low-velocity pores are prone to clogging by deposition and interception, which decreases the flow through these pores and causes higher flow rates through more open channels. Thus increased precipitate formation may result in short circuiting before clogging a bed in some cases. Corrective measures for short-circuiting may require removal of the BCR contents and rebuilding.

MIW is a challenging fluid for several conventional construction materials. Acid attacks and oxidizes most commonly used metals except perhaps for brass. Also most MIW has a significant concentration of sulfate which in concrete causes surface cracking and expansion, possibly resulting in long-term failure. Engineers should consider other potential problems, for example sulfide generated in reducing conditions plates onto metal surfaces (including 316 stainless steel) and can foul impellers. Polyurethane pumps are not prone to the sulfide plating problem; however, these pumps can be attacked by the reducing conditions and sulfide generation in addition to the acidity. On the other hand, many polymers are resistant to components of MIW, and metals and concretes can be treated to resist chemicals in MIW. The characteristics of the MIW should be checked against the properties of the planned materials.

In almost every case, the basins comprising the treatment system must isolate the water undergoing treatment from groundwater. Section 7.0 illustrate some of the technical regulatory challenges for the creation of passive treatment systems. One challenge is that the treatment system, even if built on mine tailings, may be considered a wastewater treatment system and thus must have impermeable containment.

Other components of the treatment system that should be protected from potential corrosive conditions from acidic MIW include the protection of influent and effluent structures and an instrumentation monitoring system. Influent and effluent structures should also be resistant to acid attack. Electronics should be protected from MIW, and if inside a building, electronics should be placed within a sealed enclosure and wiring should be placed in conduit piping to minimize contact with any sulfide gas. This separation can be accomplished by placing electronics in a separate space, possibly vented, with communication of wires through a conduit, which is then filled with a sealant. Specify that any interior components must have good resistance to weak (5 – 10%) sulfuric acid solutions. Although the effluent should not be as corrosive as the influent, apply similar considerations for any instrumentation required at the effluent.

4.5 Site Control Design

The BCR design should incorporate elements that minimize the potential impact of visitors. Such impact minimization has two components: hiding the structure and restricting access. If the BCR does not look interesting, it will attract r fewer visitors. For example the system shown in Figure 1-1 appears to be odd farm ponds, with little evidence of technology. People are less likely to visit such a system than one with more evidence of technology or structures. One effective method to minimize the appearance of a BCR is to design a subsurface feed into a gravel or organic (for instance, wood chip) layer so that the surface layer is all that is visible.

In addition to minimizing interest for people, also minimize the site's interest to animals. Many animals are attracted to open and running water. Covering basins such as aerobic polishing cells is generally not practical, but fencing may help to prevent entry. When the BCR system includes active components such as pumps or chemical feed, additional precautions are both advisable and required.

Access to a site is generally restricted by fencing, placing a locked gate on the access road, and placement of clear signage. In conjunction with minimal visibility, these site controls are likely to be more effective if they are low-key. Signage can prohibit trespassing but also, at the access gate, describe the project and provide emergency contacts as mentioned in Section 6.2.1.

Consider a variety of alarms to notify operation and maintenance personnel. In remote or inaccessible locations, it may be wise to duplicate alarm function to avoid unnecessary travel. Examples of alarms are: low or high water level, power loss, temperature exceedances (such as for motors) and intrusion, as well as status or alarm signals provided by any system component with a PLC or similar control. Alarms are generally delivered by an auto dialer, and newer auto dialer systems are able to deliver e-mail and text messages in addition to phone calls.

4.6 Submittals

Submittals play a critical role in construction projects and are typically required for commercial or publically funded projects. The submittal process provides protection for owners in assuring that the plans and specifications are clearly understood by Contractors building the project and determine how the Contractor intends to execute project requirements. The purpose of the submittal

process is to ensure quality, functionality and compliance with the plans and specifications prepared by the designer/engineer. In general, the purpose of submittals is to:

- Identify products selected from acceptable products in material specifications.
- Detail how items are constructed and interface with adjacent construction.
- Identify optional characteristics.
- Schedule types of products by location.
- Describe physical and performance criteria.
- Certify that products meet or exceed specified requirements

As part of the process, a Contractor is required to submit a contract submittal list at the beginning of the project. This should include all the submittals that are identified in the specifications.

The primary responsibility for scheduling, management adequacy, accuracy and control of submittals lies with the Contractor doing the work. It is important to note that approved submittals become contract documents. Therefore, the Contractor must understand that the handling of submittals is as important as the actual construction and installation activities. The construction Contractor is responsible for obtaining materials and equipment as specified in the contract.

There are three basic categories of submittals:

1. Preconstruction submittals
2. Construction submittals
3. Closeout submittals

4.6.1 Preconstruction Submittals

Preconstruction submittals are processed prior to start of construction and could include such items as:

- design basis report (bench and pilot study results)
- engineer's design report
- 30% design
- 90% design
- final design plans and specifications
- insurance certificates
- payment and performance bonds
- subcontractor and products lists
- construction schedule
- federal, state and local permits (see http://cfpub.epa.gov/npdes/indpermitting/mining.cfm)
- submittals schedule
- schedule of values
- erosion and stormwater control
- environmental protection, health and safety, construction quality assurance, pollution, and traffic control plans

Each owner may have different requirements for the different preconstruction submittals. For example, payment and performance bonds may not be necessary for small projects and could be waived. Traffic control plans may not be required for remote locations with little traffic, but would be required at sites with significant equipment and vehicle/truck traffic such as at active mining sites.

4.6.2 Construction Submittals

Construction submittals are processed during construction and could include such items as:

- shop drawings
- coordination drawings
- product data
- samples
- quality assurance/quality control (QA/QC) submittals
- informational submittals
- construction photographs
- sustainable design submittals
- payment applications

The type and amount of construction submittals will vary based on the project and documentation requirements. The Contractor is typically required by the contract documents to submit shop drawings, product data, and samples for certain parts of the work. Per the American Institute of Architects, "shop drawings are drawings, diagrams, schedules and other data specially prepared for the Work by the Contractor or a Subcontractor, Sub-subcontractor, manufacturer, supplier or distributor to illustrate some portion of the Work." The owner's representative is typically required to review and approve or take other appropriate action upon the receipt of the Contractor's submittals such as shop drawings.

The principal reason engineers review shop drawings is to evaluate whether the contractor understands the engineering design concepts and to correct any misinterpretations before they are constructed. If the shop drawings indicate that the work depicted will not comply with the intent of the contract drawings and specifications, the engineer has an opportunity to notify the Contractor before the costs of fabrication, purchase, or installation have been incurred. The costly and wasteful alternative to this procedure would be simply to wait until the work has been constructed, inspect it and then either approve or reject it. It is much more economical to review and correct the shop drawings than to remove and replace erroneous construction.

4.6.3 Closeout Submittals

Closeout submittals are processed as part of project closeout and could include such items as:

- written notice of substantial/final completion
- as-built drawings

- final application for payment
- record documents
- O&M data and manuals
- spare parts and extra stock materials
- certificate of payment
- release of liens and waiver of debts and claims
- consent of surety to final payment
- special warranties/Warranty Management Plan
- keying
- sustainable design submittals

As with the construction submittals, the type and amount of closeout submittals will vary based on the project and documentation requirements of the Owner. Typically as-built drawings are required to provide documentation of what was actually constructed and provides the owner with a permanent record of each project feature.

Operation and maintenance data manuals is a critical closeout submittal and includes such items as operating instructions, safety precautions, preventative maintenance, troubleshooting guides, spare parts and supply lists, and warranty information.

5.0 CONSTRUCTION

BCR performance depends on the care exercised during construction. Much of the construction is accomplished using heavy equipment, often in areas with poor or limited access, and by workers who may be more accustomed to less stringent design parameters. This chapter discusses the construction quality assurance (CQA) necessary during BCR system construction. Construction specifications, developed during the design phase of the project (Section 4.6) describe acceptable products and material specifications, alternatives, performance criteria and schedules to be followed by the construction contractors and confirmed by the site construction engineer.

Following the discussion of CQA are description of the construction features at a site using a BCR to treat MIW. These features include the following:

- the physical setting of the BCR systems (Section 5.2.1)
- typical BCR construction elements including site preparation (Section 5.2.2.1), liners (Section 5.2.2.2), process piping (Section 5.2.2.3),substrate installation (Section 5.2.2.4), cover installation when necessary (Section 5.2.2.5), and effluent control (Section 5.2.2.6)
- system performance and monitoring equipment
- site construction restoration
- site security

As a reminder, health and safety requirements during construction, operation and maintenance of BCR systems include those typical of remote construction projects that are also regulated by Occupational Safety and Health Administration (OSHA) and the Federal Mine Safety and Health Act, administered by the Mine Safety and Health Administration (MSHA), as applicable. In many instances training on excavation activities, confined space entry procedures, work with heavy machinery, power tools, uneven terrain, power tools, and work at and near contaminated sites can require 40-Hour Hazardous Waste Operations (OSHA HAZWOPER) training for those workers that perform activities that expose or potentially expose them to hazardous substances during construction as well as during the treatment of hazardous substances.

Other regulations may apply to the siting of BCRs, including the Endangered Species Act of 1973 which protects the habitats of endangered species. On other sites artifacts or structures with cultural or historic value may be protected by The National Historic Preservation Act of 1966. In addition construction permits may be required from tribal, state, or local authorities. Please consult federal, tribal, state, and local requirements prior to construction.

5.1 Construction Quality Assurance (CQA) Plan

A Construction Quality Assurance Plan (CQA Plan) describes the construction quality assurance procedures in use during the BCR system construction activities. This plan is a recommended project submittal for the BCR based on the design requirements.

The CQA Plan addresses construction quality assurance procedures that are to be used to independently test and verify that the construction activities meet the intent of the design and conform to the construction specifications and drawings. The CQA Plan may be created by the Engineer as part of the design or by the Constructor as defined by the construction specifications. Example components of a CQA Plan are presented below, along with descriptions of key elements of a CQA Plan including definitions, construction quality control (QC) protocols, design modifications, clarifications, and changes, certification reports, and surveying.

5.1.1 Example Table of Contents for a CQA Plan

Table 5-1 presents an example table of contents for a CQA Plan applicable to construction of a BCR system. This example is not intended to be a definitive list of components of a CQA Plan as site-specific design may require different or alternative components.

Table 5-1. Biochemical reactor system CQA plan table of contents (example)

1.0 INTRODUCTION
1.1 Definitions
1.2 Coordination
1.2.1 Pre-Construction Meeting
1.2.2 Progress Meetings
1.2.3 Documentation
1.2.3.1 Log Books
1.2.3.2 Daily Construction Reports
1.2.3.3 Photos
1.2.3.4 Receipts and Weight Tickets
1.2.4 CQA Reports
1.2.5 Design Modifications, Clarifications, and Changes
1.2.6 Certification Report
1.2.7 Surveying
1.2.8 CQA Schedule of References
American Society for Testing and Materials (ASTM)
Geosynthetic Research Institute (GRI)
U.S. Environmental Protection Agency (USEPA)
2.0 CONSTRUCTION QUALITY CONTROL
3.0 CONSTRUCTION QUALITY ASSURANCE
3.1 Earthwork Inspections
3.2 BCR Plant Growth Medium Cover Placement and Thickness
3.3 Piping, Drainage Gravel, and Geotextile Materials for BCR Drainage Layer
3.4 Concrete Work
3.4.1 General
3.4.1.1 CQA Personnel
3.4.2 Materials
3.4.2.1 Submittals

3.4.2.2 Quality Assurance Conformance Testing
3.4.3 Inspections
3.5 Oversight of Seeding and Hydro-Mulching Operations
3.6 Geotextile Installation
3.7 Geomembrane Installation
3.7.1 Geomembrane type 1
3.7.2 Geomembrane type 2
3.8 Electrical and Instrumentation Installation
ATTACHMENTS
Attachment 1. CQA Forms
Attachment 2. Geomembrane Liner Material Specifications

5.1.2 Key features of the CQA Plan

5.1.2.1 Definitions

The following sections include example definitions for terms used in a CQA Plan. These definitions might be customized for different projects.

Construction/Project Manager – Referred to as the Construction Manager or Project Manager. This individual is the official representative of the Owner and is responsible for all construction activities, including oversight and direction during construction. The Project Manager is also responsible for coordinating construction and CQA activities for the project. The Project Manager serves as a liaison between all parties involved in the project to ensure that ongoing communications are maintained and that resolution is reached for all CQA issues that arise during construction. Duties for the Construction/Project Manager may include:

- Review construction drawings and project specifications.
- Coordinate with the Engineer / Engineer of Record prior to construction to ensure that the construction drawings and specifications are integrated with the CQA program.
- Administer the construction contracts, including any associated changes that occur during the course of construction.
- Assure that Engineer of Record receives required information for review, change requests and other information to assure completion of certification report.
- Coordination of all construction activities
- Coordination with regulatory agencies
- Coordination and implementation of project Health and Safety Plan
- Scheduling and coordination of construction activities with required CQA testing and activities

Engineer – Also referred to as Design Engineer or Engineer of Record, this individual is responsible for the design and preparation of the construction drawings and specifications. The Engineer should be a registered professional engineer in the state in which the work occurs. The Engineer should have the authority to stop any aspect of the work that is not in compliance with the CQA Plan. Work would then resume once a corrective action has been approved by the Engineer. In the

absence of the Engineer from the work site, the duties and responsibilities of the Engineer are delegated to the CQA Monitor(s). The specific responsibilities of the Engineer may include the following:

- Develop the construction drawings and specifications, and CQA Plan.
- Train CQA personnel on CQA requirements and procedures.
- Schedule, review, coordinate, and approve CQA activities to ensure that testing and documentation are complete, accurate, and in accordance with the site-specific design requirements.
- Review the QC operations performed by the Contractors(s).
- Ensure that required CQA testing has been performed in accordance with this CQA Plan as well as the intent of the construction drawings.
- Confirm that test data are accurately recorded, validated, reduced, summarized, and interpreted.
- Approve specific corrective measures to be implemented during construction when deviations from the design and this CQA Plan occur.
- Follow Project Manager-approved Request for Information (RFI) and change order procedures.
- Develop a project file to maintain and store originals or copies of originals of all CQA data sheets and reports generated during the course of construction and following completion of the project. A complete file of this documentation is also maintained by the CQA Monitor.
- Oversee preparation of the Certification Report at the completion of the project.
- Provide final certification that the BCR system has been constructed in accordance with the construction drawings and specifications, and related project documents.

Construction Contractor– The group(s) responsible for performing the earthwork, construction, revegetation, and other construction activities associated with the BCR system. The Construction Contractor may employ various subcontractors, approved by the Owner and Engineer, for completion of the work.

Construction Quality Assurance Monitor – Also referred to as the Monitor or CQA Monitor, this is the firm or individual responsible for performing the tasks outlined in this plan. The CQA Monitor reports directly to the Engineer. The following may be included in the CQA Monitor's duties:

- Review and understanding of all construction drawings, Contractor-provided shop drawings, project specifications, and related guidance documents.
- Review any required Contractor submittals and make appropriate recommendations.
- Observation and documentation of sub-grade preparation, compaction, cover material placement, surface water controls construction, planting materials and methods, and other site work activities.
- Ensure that testing equipment used and tests performed are conducted according to project specifications and industry standards.

- Coordinate development of the Record Drawings.
- Perform or observe documentation and reporting of test results as required to Engineer and Project Manager.
- Report to the Engineer any deficiencies that are not corrected to the satisfaction of the CQA Monitor, including design or project specification changes.

Owner – Company or Agency that pays for the work to be completed.

Project Construction Drawings and Specifications – Includes all project-related construction drawings and specifications, including design modifications, specification changes, and Record Drawings.

Quality Control (QC) – Functions performed by the Construction Contractors or supplier to conform to the construction drawings and specifications. The Construction Contractors and its designated consultants should implement QC procedures to monitor construction methods, survey controls and any other controls as needed by the Construction Contractors to meet the project specifications and requirements.

Record Drawings – Record (as-built) Drawings record the dimensions, details, and coordinates of the final grading, surface water controls and other structures after construction is completed. The Record Drawings may show any design or field modifications or changes from the original design Drawings.

5.1.2.2 Construction Quality Control

Construction quality control functions are performed by the QC Monitor(s) during construction. The role of the QC Monitor, who is a representative of the Construction Contractors, is to report to the Construction Contractors and to provide construction support and controls so that the design specifications are achieved. The QC Monitor also must coordinate with the CQA Monitor and Engineer.

The Construction Contractors is responsible for developing the QC methods and procedures that are used in the construction, and ensuring the methods are consistent with the construction drawings and specifications.

The Construction Contractor performs and keeps a record of all visual and physical checks. Visual checks (such as material integrity, seaming, and overlaps) may include photo-documentation as well as observation notes to document performance and compliance with the Drawings and Specifications. Physical checks involve surveys, grade checks, physical measurements (such as for proper overlaps, lift thicknesses, and structure dimensions) or other data collection activities, as required to document compliance. Examples of the primary QC functions and observations include but are not limited to the following:

- clearing
- keeping inventory of equipment, materials and personnel on site
- testing fill support to CQA (and associated QC testing and surveying)
- surveys for grade control and for submittal and review by Engineer to review for conformance with design requirements and tolerances
- QC testing
- excavation, and fill placement for conformance to Drawings
- proportioning and mixing of sulfate reducing bioreactor organic substrate
- application of bacterial inoculum on the BCR unit surface if manure is not used as seed
- installation of pipelines and associated plumbing for burial depth conformance
- installation of electrical components including power, instrumentation, and alarm systems for functionality conformance
- testing of riprap or other fill materials for conformance with gradation and physical property requirements
- surface water survey controls (e.g. channel construction, conformance with gradient and cross-section design tolerances, erosion protection revetment)
- concrete structures, including performing QC testing of concrete in accordance with the specifications
- BCR cover placement and thickness control (grade stakes or pre-and post-survey that account for potential lightweight fill (LWF) settlement)
- plant growth medium preparation and placement
- seedbed preparation and seeding
- installation of wetland plants in the aerobic polishing cell (APC)
- record surveys
- submittals to the Engineer for review (for instance, shop drawings and RFI forms) and inclusion in Certification Report

Prior to installation of any materials, the Construction Contractor should verify that all construction materials meet the required specifications. The Construction Contractor should provide the CQA Monitor and Engineer with the manufacturer's QA/QC documentation including, at a minimum: a certificate of analysis for each lot of material supplied; a description of the manufacturer's QC procedures for verifying final product quality; product identification numbers; and results of standard material properties testing verifying that the materials meet minimum material requirements.

Any substitutions for the specified materials should be requested in advance and receive approval from the Owner and Engineer prior to use. Substitutions should meet the minimum requirements for the specified material for approval. The Construction Contractor is responsible for all materials from the point of shipping until the material is placed consistent with the BCR design specifications. The CQA Monitor should visually inspect, inventory and approve all materials upon delivery.

The Construction Contractor should assist the Engineer and the CQA Monitor with the test fills to attain the required method specifications for compaction. The test fills QC requires survey control per standard Proctor (ASTM D-698) compaction testing.

5.1.2.3 Design Modifications, Clarifications, and Changes

During construction, the need to document design and/or specification clarifications, modifications, and/or changes may arise. In such cases, the CQA Monitor should notify the Construction Manager and the Engineer of the need for any clarifications, modifications, or changes. This notification usually takes the form of a written agreement with documentation by the Engineer and acknowledgment by the Construction Manager. All design deficiencies and construction problems should be documented as well as any design modifications that resolve the issue. Documentation should generally include but not be limited to:

- detailed description of the problem or deficiency including location, cause, how, and when the deficiency was identified
- how the situation or deficiency was resolved
- any measures taken to prevent similar problems in the future
- signature of the CQA Monitor, Construction Manager, Construction Contractor, and certification by the Engineer
- P.E. certification by Engineer for any design changes

5.1.2.4 Certification Report

At the completion of the project, the Engineer should submit a Certification Report to the Project Manager and the Owner. This report certifies that the construction work was completed in compliance with the construction and shop drawings, project specifications, and related documents and be stamped by a registered (state) Professional Engineer. The report should include but not be limited to:

- summaries of all construction activities
- observation and test data sheets
- sampling locations, tests performed, and test results
- description of any significant construction problems and/or design deficiencies and how they were resolved
- copies of any design and/or specification clarification, modification or change forms
- documentation submitted by the CQA Monitor (daily construction reports, weekly progress reports, pictures, and field test results)
- record (as-built) drawings that accurately locate all site work such as construction items, erosion control systems, dimensions, and cover thickness

5.1.2.5 Surveying

Record survey documentation should be conducted as part of the Construction Contractor's QC, with the CQA Monitor conducting periodic spot reviews and checks. The use of a Global Positioning System (GPS) receiver calibrated to the local site GPS network using site coordinates can assist in this documentation.

5.2 BCR Construction

5.2.1 Physical Settings

5.2.1.1 Siting the BCR System

Most passive treatment systems are located in remote, hard to access areas close to the source of MIW to treat. In some instances the mining-influenced water is captured and conveyed, piped, or pumped to the BCR system, which may be some distance from the MIW source.

Although the design of the BCR already considers numerous site-specific construction factors, finalizing the protocols, timing and procedures to bring construction equipment, materials and personnel to the BCR construction site is recommended. Constructing the BCR system during the early spring sometimes is recommended to avoid unnecessary vegetation clearing and avoid construction during rainy and winter seasons. Commissioning of BCR systems during the summer and fall are recommended. The local vegetation of the area takes a significant role where the influent water is routed to a convergence point because the plants can increase the dissolved oxygen (DO) of the MIW and may cause some metals to precipitate.

5.2.1.2 Water Influent Control System Installation

The simplest BCR water influent conveyance is the one that occurs naturally as a single stream that connects the MIW to the BCR system. The construction phase may call for increasing the stream side banks to match a specific maximum design flow that the BCR can support and channeling the water flow to the BCR influent structure. During intense precipitation events the maximum design flow may be surpassed and by-pass or diversion channels need to be constructed to relieve the excess flow to the BCR system. It is common to construct an influent bypass during construction and/or in high flow conditions. A by-pass box can be designed such that the excess overflow is diverted hydraulically using an overflow weir to the by-pass system.

In many cases the design includes an influent control structure (see Section 4.4.2.1). The Construction Contractor determines the construction methods for this structure, including the method applied to bypass MIW from the site during the construction process.

In other cases, the capture and conveyance of the MIW influent requires the design and construction of permanent structures such as spillways, or underground piping that connect the influent flow to the BCR system. Figure 5-1 show one example influent structure that required the use of an underground concrete piping that directs the influent to the head of the BCR system.



Figure 5-1. Example of influent structure on a BCR.

The presence of sediment, either in the MIW influent or resulting from the BCR construction phase, can negatively affect the performance of the BCR and must be managed and controlled using temporary or permanent settling basins or sediment traps.

In addition to MIW local regulations, tribal, state, and federal regulations may apply to storm water management at the site during construction. Stormwater management during construction of the BCR can be as simple as inspecting the stormwater drainage area or basin to ensure that minimum stormwater flows are allowed to the BCR system. In other cases, major stormwater drainage basin modifications would be needed and called for in the design. Implementing these changes may require close coordination with local and state watershed basin stakeholders. Some of the storm-water management practices that are implemented during construction, specifically those aimed at managing soil erosion and controlling the disturbance of sediment and siltation, are also imple-mented during operation of the BCRs.

5.2.2 BCR Typical Construction

This section presents guidance on typical construction aspects of a BCR unit, including site pre-paration, liner installation, process piping installation, substrate installation, and cover installation. Although the overall treatment system may consist of the BCR unit, influent and effluent com-ponents, and pre- and post- treatment components, this section focuses on construction of the BCR unit component, and also presents general discussion of the relation of the BCR unit construction to other components of the overall treatment system.

5.2.2.1 BCR Site Preparation

Construction of a BCR system begins with preparing the site area. The site preparation task for the BCR unit includes, but is not limited to:

- road access improvements and physical hazard markings
- development of staging and temporary facilities
- development of erosion control and storm water runoff features
- clear and grub of trees, stumps, and brush
- earthwork preparation of the BCR unit base and other related components
- management of excavated and cleared and grubbed materials

Abandoned or active mine sites often have disturbed land areas in which the BCR can be constructed. In this case, minimal clear and grub activities may be required. Many mines are also located in steep or sloping terrain. In these cases, areas must be excavated and re-graded according to the design. In remote areas, road construction may be required as the first scope of work in order to provide access to the site. Additionally, areas disturbed by mining activities may be contaminated by metals or other constituents. If contaminated materials are present, excavated materials must be handled in accordance with the design or other site requirements, such as disposal in an on-site repository or off-site location.

BCR units can be in-ground or above-ground, depending on site conditions. For in-ground units, the unit area is excavated according to the construction plans, and overburden (soil) material may be stockpiled for construction of other components of the overall treatment system. Above-ground units require borrow from a stockpile of overburden already present on site, developed from a nearby area, or brought in from off site in order to construct the unit berm. Some BCR units may also be partially in-ground, where the excavated overburden is used to build a berm on the sides of the unit, thereby extending the overall depth of the unit. It is also possible that a BCR unit is constructed within an area of bedrock, in which case bedrock must be blasted and removed to create the depression necessary for the unit installation.

If deemed acceptable in accordance with the design, excavated overburden material could also be used for construction of several other components such as run-on/run-off control features (such as berms or drainage channels), riprap, influent/effluent channels, pond berms, roads and other access features, growth media for vegetation and erosion control, and cover material for the BCR unit. Importing off-site material for construction may also be required for these various components. Reuse of excavated overburden or other local on-site resources is an important component of the project, as this can reduce overall cost of hauling materials. Since these various uses may require different particle size gradations and/or other soil/rock properties, overburden may need to be screened, separated, and amended in accordance with the design. There is also potential to chip or shred the cleared and grubbed woody vegetation material and incorporate it into the BCR unit substrate, if this material is deemed appropriate as prescribed by the design.

The design determines what site preparation procedures are to be followed. Construction QA/QC procedures must be adhered to during site preparation to ensure the BCR unit size, depth, and grades meet the design requirements, which are critical components for successful operation of gravity fed systems. Surveying should be conducted prior to, during, and after unit construction to ensure the line grades are met. The elevations of the influent, effluent, overflow, and other features must meet the design requirements in order to reduce risk of future potential issues with improper loading to the system. Issues that could result from poor construction grading could include overloading the system beyond the designed limits and allowing the system to become dried out. These system stresses can lead to issues such as reduction in treatment efficiency, loss of appropriate bacterial community, and plugging as further discussed in Section 4.0.

Another element of site preparation construction is implementation of proper erosion control practices. As common for earthwork construction, materials such as silt fences, diversion channels and ditches, sediment capture ponds, and similar features may be required. The main purposes of utilizing erosion control practices are to prevent damage to downstream resources, limit loss of material stockpiles or other features as a result of stormwater erosion, and to route stormwater around construction areas to improve the safety of work conditions. For the BCR unit, it is also important to have a storm water diversion and dewatering plan in place in the case of high precipitation events occur during construction. Proper implementation of these features is typically required prior to all land disturbance. An erosion control plan and/or storm water pollution prevention plan should be developed and implemented that defines the approaches for erosion controls.

Since the intent of the BCR system is to treat MIW, an actively flowing source of MIW may be present near the BCR construction area. This MIW should be managed accordingly to prevent it from entering the work area and prevent contamination of otherwise clean soil or granular fill resources used for construction. MIW should be routed around the work area, treated, or stored in a containment pond as required per the design, mine operating plan, or other requirements.

5.2.2.2 Liner Installation

The base or floor of the BCR unit may be constructed in a number of different ways depending on site conditions. In the majority of cases, a liner is specified to prevent seepage of surrounding groundwater into the BCR unit or leakage of MIW from the BCR unit. In the case that the BCR unit is constructed within native clay or similar low-permeability soil, a liner may not be required. However, adequate testing of the soil conditions should be conducted to ensure that the BCR unit holds water.

A liner system can be constructed of a number of different materials that are common for dam, pond, or impoundment construction (see Section B.13). These material types may include PVC, high density polyethylene (HDPE), linear low density polyethylene (LLDPE), and geosynthetic clay liner (GCL). For more information on liner components see *Use of Geosynthetics in the Mining and Mineral Processing Industry* (http://geosyntheticsmagazine.com/articles/0807_nags_processing.html). These materials are usually laid in place in sections, and sections are heat welded together in the field to create an impermeable seal. Geotex is generally placed over liner prior to

gravel placement. Installation of these liner materials must follow strict CQA procedures to ensure that the permeability and other material properties are met as prescribed by the design. These procedures include but are not limited to:

- inspection of liner material for defects
- documentation of sheet material properties and testing to meet the design requirements
- evaluation of proper liner bedding material to prevent liner damage or puncture
- field trial seam development and testing to determine if the field welding process creates seams that meet the design requirements for material properties
- destructive and/or nondestructive field seam testing ensure the continuity and integrity of in-place field seams

For some materials, the liner can be manufactured as one continuous piece. In this ideal situation, field welding of sections would not be required, thereby reducing the amount of construction cost associated with welding and testing. The only field welding that may be required in this case is for pipe penetrations for the influent and/or effluent piping.

After completion of the of the site preparation for the BCR unit, the liner bedding is installed. The bedding material is typically a non-angular sand or fine gravel so as to not damage the liner. Due to the specific particle size and shape requirements for bedding material, it may be imported from an off-site location that has the appropriate processing equipment. Alternatively, on-site resources may be present and could be used. As described for site preparation, use of on-site materials for bedding can help reduce costs. The bedding layer is placed according to the thickness defined by the design (typically 3 to 6 inches) and compacted. The bedding surface should be smooth and free of rocks or other protrusions that can damage the liner. Inspection of the bedding should occur prior to liner installation to ensure the design conditions are met.

The liner is installed after placement and inspection of bedding. Liner installation can proceed by first anchoring one side of the liner sheet into the pond berm, and rolling the sheet across the BCR unit base. The liner may also be laid out over the BCR unit with excess liner temporarily anchored until field welding is completed. Ultimately, the method of liner installation should be per the manufacturers' and experienced installers' recommendations. Final anchoring of the liner on all sides of the BCR unit is conducted by excavating a shallow trench (typically 1 to 2 feet deep and up to 1-foot wide), laying the liner along the walls and floor of the trench, and backfilling and compacting soil into the trench. Backfilled trench materials in direct contact with the liner should meet the same typical requirements as for liner bedding.

Field seaming of sheets is conducted in accordance with the design and manufacturers recommendations and should be conducted by qualified personnel. Penetrations through the liner for influent and/or effluent piping may be required. A pipe penetration is installed by cutting the proper size hole in the installed liner, feeding pipe through the penetration, welding a piece of liner material to the sheet around the pipe, and clamping the liner around the pipe to create a seal.

The last component of liner installation is field seam testing of sheets and pipe penetrations to ensure that leakages do not occur and to test the properties of a field seam. Field seam testing can be conducted using destructive or non-destructive methods, depending on the type of material installed, extent of field seaming, and field conditions. Destructive field seam testing involves collecting a sample of the seamed area and submitting to the laboratory for testing. Non-destructive field seam testing involves either vacuum testing (pulls a vacuum on a seemed area) or air pressure testing (inserts air into the enclosed space between a double-track fusion seam). The type of testing method to be used is dependent on the type of liner material and type of field seaming method. Based on the results of field seam testing, repairs to seams may be required, and the area should be re-tested upon completion of the repairs.

5.2.2.3 Process Piping Installation

Process piping for BCR systems includes influent and effluent piping outside of the BCR unit, and piping within the BCR unit. As for site preparation tasks, it is critical to install process piping to meet the grading, pipe material type, and pipe connection requirements prescribed by the design. These elements are critical to success of a long-term functioning system. Typical process piping material for BCR systems are PVC, smooth-walled HDPE and corrugated HDPE. Within the BCR unit, PVC or corrugated HDPE piping is most common because of lower price and ease of installation. Effluent collection piping for a downflow BCR unit is also typically perforated. Effluent for an upflow BCR unit may not require effluent piping, rather the system is graded to drain by overflow at the top of the unit although during high flow conditions the potential for short-circuiting at the BCR may be observed if not designed properly.

For downflow BCR units, effluent piping is installed at the bottom of the unit. Pipe bedding is first installed over the liner to aid in achieving proper grade. Pipe bedding in direct contact with the liner system should be similar to the bedding used for liner installation to prevent liner puncture. Bedding directly around the perforated pipe is commonly a poorly graded gravel to aid in drainage. This material may also be limestone gravel, which can add additional alkalinity to the effluent. Gravel or any other backfill placed over piping should be conducted carefully to prevent damage to the piping system. The depth of gravel over top of the effluent piping should be placed in accordance with the design and confirmed in the field prior to additional backfill of substrate into the BCR unit.

The effluent piping can be configured in a number of ways. For example a gallery of perforated effluent piping may be installed. This consists of several lines of perforated pipe in parallel which are connected a main effluent header that feeds to the effluent pipe through the liner penetration. Assuming the effluent pipe through the liner is installed near the bottom of the BCR unit, effluent piping installation should continue from this point outward, or the gallery should be constructed, graded, and connected to the effluent pipe. All piping, couplings, elbows, and other fittings should be inspected for defects prior to installation. The effluent system may also contain more than one effluent pipe gallery in parallel. In either case, grade checks should be conducted throughout effluent pipe installation to ensure grades are met. All inspections of grade should be documented on

appropriate forms or in a logbook. Adjustment to grade can be achieved by adding more or less bedding beneath the piping.

For upflow BCR units, influent piping that evenly distributes the flow to the bottom of the system would be installed similarly to that of an effluent collection piping gallery described above for a downflow system. The key difference would be proper pipe grading in the opposite direction of the effluent piping for a downflow system. For an upflow system, the effluent would typically rise to the surface and overflow into discharge channels or other flow control structures such as an overflow spillway.

Some horizontal flow BCRs have been constructed, but are not as common as downflow BCRs. Flow can be delivered to a horizontal flow system along one side of the BCR unit via a series of influent pipes or a perforated pipe gallery that evenly distributes flow. Similarly, effluent would be drained on the opposite end of the unit using a collection gallery. Perforated influent or effluent galleries would be constructed similar to upflow or downflow BCRs, expect that the piping galleries may be raised off of the unit floor. Suspended piping systems may require adequate anchoring and support systems during construction. In other cases, the flow follows a serpentine pattern through multiple reaction units where the influent to one unit enters at the bottom at one corner and the effluent leaves at the opposite corner of a rectangular or square reactor (see Section B.10), as dictated by space availability.

Piping installed outside of the BCR unit also requires bedding material, although the type of bedding is typically a well-graded sand or gravel, or alternatively on-site overburden materials could be used. Trench excavation would be required for buried piping. Bedding is placed within the trench, and then pipe is installed to grade. Backfill is placed over the piping and compaction is completed with caution to not damage the piping. A common practice to install flagging or a warning tape at least six inches above the piping to prevent potential damage if piping ever requires future maintenance. Above ground piping may require anchoring systems, especially on steep slopes.

Valves are not likely within the BCR unit, but may be found at the effluent or influent of the unit in areas that can be accessed for maintenance. Construction of access to valves or other process control equipment may require construction of an in-ground manhole or vault, and require additional safety precautions during operation of the BCR. Installation of these kinds of structures must also occur with close attention to required grades. Proper bedding for structures should be installed, compacted, and tested for density as required. This bedding is critical to reducing potential for future settling of the structure, which could lead to damage to piping entering or exiting the structure. Below ground structures may also be considered confined spaces and adequate procedures should be implemented to ensure the safety of workers during installation and during future operations and maintenance. Sampling ports or automated monitoring devices may also be present within the structure or near the valve area. Connections to valves, or any other device, to process piping should be conducted by qualified personnel to ensure leakage does not occur. Leak testing of connections or the piping system, as a whole may also be required prior to backfill over piping. All testing should be documented on appropriate forms or a log book.

5.2.2.4 Substrate Installation

As discussed in detail in Section 3.1, the substrate for a BCR can include a number of different materials. The substrate in a BCR is the main component by which the MIW is treated. Typical BCRs include solid substrate materials placed in the unit. In some cases, liquid substrates may be added to the influent stream. The types and proportions of materials are developed in the design based on results of the bench and pilot testing that indicate the best treatment approach for the site MIW. As such, the proper proportioning of substrate materials for the full-scale system is vital to the success of the reactor to adequately treat the MIW. If not done during the testing phase, the substrate materials should undergo comprehensive testing for incidental contaminants that could be introduced into the system.

Installation of the BCR substrate begins with understanding the conversion between volumetric and mass quantities of substrate materials. Bench and pilot studies and design parameters may be developed using either one or the other type of quantity, or both. Purchase and hauling of large quantities of substrate materials may also be measured in either unit. Therefore, the conversion between these units (density) is critical to properly proportioning the substrate materials. Additionally, the percent moisture can have a large effect on materials measured by weight. Prior to construction (if not done during the design), samples of substrate materials can be submitted to a laboratory to test the density and percent moisture of the materials (Section 3.2.1). This step may be conducted during the bench or pilot test phases. Alternatively, this information may already be available from the material supplier or determined based on past experience and/or literature values at the discretion of the Engineer and Construction Contractor. The material properties are then used to properly coordinate with the Construction Contractor on how to blend the materials in the field.

Delivery of each of the substrate materials to the site in truckloads should be placed in separate areas prior to field mixing. Upon each delivery, documentation of the contents volume and properties should be provided in accordance with the CQA Plan. Based on the proportioning of materials in the design, mixing batches of substrate can then commence. This procedure should proceed using a method that is simple using standard earthwork construction equipment. One approach is to proportion the designed substrate mix using excavator bucket loads of each material. The size of the bucket is determined and then fractional amounts of each component are grabbed with the excavator and placed in a pile. The pile is then mixed using the excavator bucket, site workers, and/or tilling equipment. Concrete mixers also have been used. Such mixers mix well but have limited batch sizing.

As part of CQA requirements, the Construction Contractor should create a test batch prior to full-scale mixing. The test batch would use the same approach as described above, but at a smaller volume. Development of the test batch should be observed by construction oversight personnel. This observation allows for calibration of the mixing method used that ensures the designed proportions are met. The mixing procedure should be approved by the oversight personnel prior to full-scale implementation.

After installation of the pipe and gravel bedding, the substrate is installed. A geotextile filter material may be required over the gravel layer prior to substrate placement. The geotextile reduces loss of fine-grained substrate material into the gravel layer, while allowing adequate water flow to the effluent drain system; however, the geotextile should not be too fine that in may cause plugging.

Placement of substrate material within the BCR unit can be conducted in a number of ways, but all should be implemented with caution to not significantly compact materials once placed. Significant compaction of substrate material can cause reduction in permeability in certain areas of the BCR unit and result in reduced treatment efficiency and potential plugging issues. In some cases, the design may call for minor compaction to meet a certain permeability requirement; however, the key for substrate placement is to provide uniform compaction throughout the unit. Non-uniform compaction or excessive compaction is some areas can cause preferential flow pathways in less compacted areas and reduction in treatment efficiency.

Substrate materials can be installed from equipment along the outside of the BCR unit, with light-duty equipment within the unit, or a combination of both methods. Placement depends on whether water flow is vertical or horizontal.

Vertical flow BCRs: For installation from the outside of the BCR unit, an excavator is used to place substrate into the BCR unit and spread the material without entering the BCR unit. This process proceeds around the entire perimeter of the unit until all material is placed and leveled to the required depth. This approach reduces the issues regarding compaction of the substrate or damage of collection pipes while the substrate is being installed. However, this method is limited to placement at the reach of the excavator. If the excavator reach is less than the geometric radius of the BCR unit, then proper spreading of the substrate cannot be conducted using the excavator. In this case spreading substrate in the center of the BCR unit must be conducted by small track mounted skid loaders that can enter the BCR. Figure 5-2 shows sketches of vertical flow BCRs.

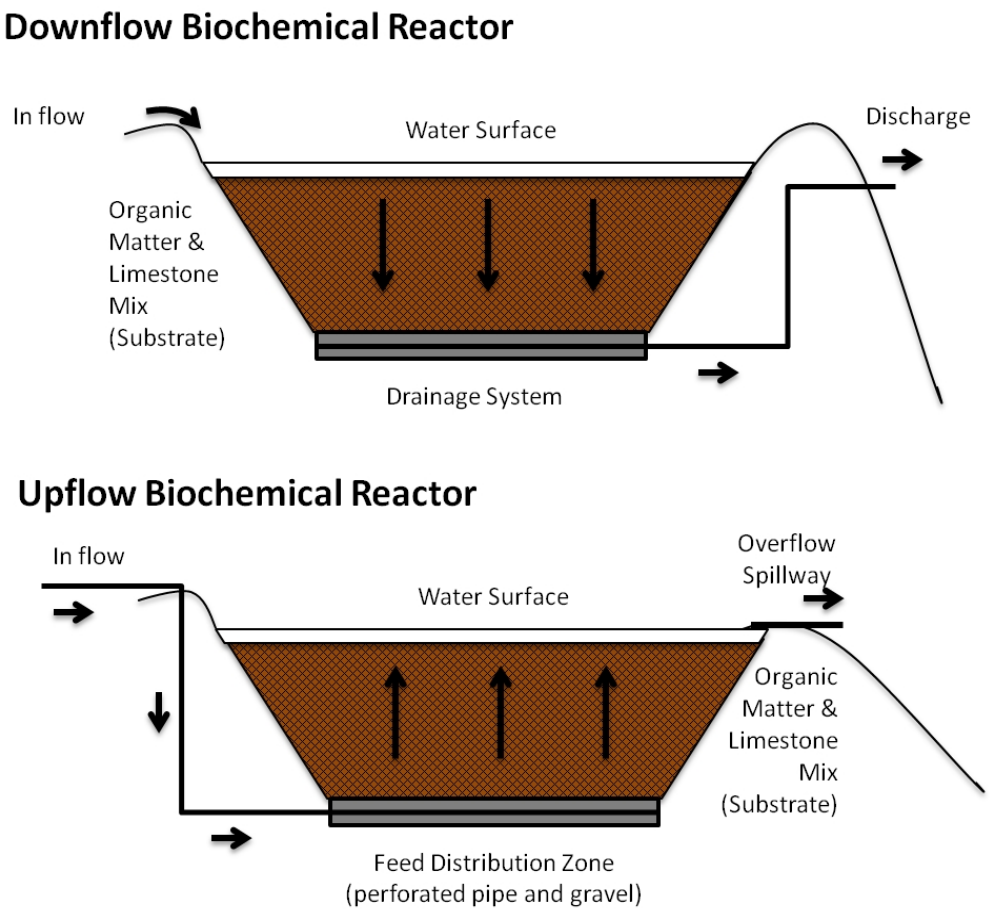


Figure 5-2. Vertical downflow and upflow BCRs.

If equipment can be operated on the gravel bedding layer above the liner, another method of substrate placement is to dump truckloads periodically spaced throughout the unit. The decision to drive trucks over the gravel bedding (portions of which may contain buried pipe) should be made based on the compaction properties and thickness of the gravel placed over the piping gallery. An entrance ramp to the BCR unit should be constructed with a layer of additional protective material over the liner to prevent damage by heavy construction equipment. A small track mounted skid loader should then be used to spread the substrate piles into a uniform depth. The loader should start at the opposite end of the entrance ramp to the BCR unit and work backwards toward the entrance until the entire BCR unit has been graded to a uniform depth.

Horizontal flow BCRs: As with vertical flow BCRs, an excavator is used to place BCR substrate and alternating rows of gravel, but the unit may be constructed from one end to the other while equipment operates on the non-filled portion of the unit. Figure 5-3 shows a sketch of a horizontal BCR, the upper of which is at the Peerless Jenny King site (see Section B.10) and the lower is an example included in the ADTI Handbook (http://wvwri.nrcce.wvu.edu/programs/adti/pdf/adti_handbook_5.pdf).

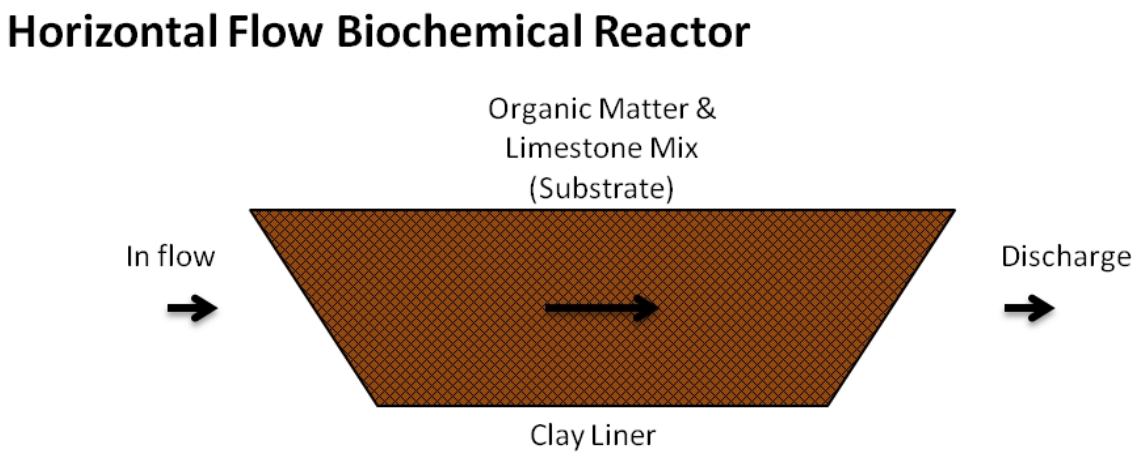


Figure 5-3. Cross-sections of horizontal flow BCRs.

The method of constructing the rows of substrate and gravel may not be specified and may be labor intensive. An illustration of one such installation is shown in Figure 5-4.



Figure 5-4. Horizontal BCR media placement.

Source: Jeff Schoenbacher Park City Municipal Corporation.

Another alternative for substrate placement regardless of flow direction is construction of a dividing earthen berm between parallel BCR units or sections of a BCR unit. If not in the design drawings, adding berms require the Construction Contractor to obtain concurrence from the Owner and Engineer. The berms provide the benefit of an access route for construction equipment to install substrate and also to perform maintenance on the substrate. The berms can be constructed within the overall lined system. As an example, berms can be constructed that divide a rectangular unit into two or more sections. The berm should be constructed after the liner installation, but prior to or during bedding and effluent pipe installation. The berm height should be equivalent to the BCR unit height at the perimeter of the unit and compacted in lifts as appropriate. In some cases part of

the top of the berm is built at a lower elevation to allow for overflow by design or serve as the actual effluent of the BCR. A protective layer of soil should be placed directly over the liner surface prior to placement of other soil materials that may contain angular gravel or other debris that could damage the liner. The segregated areas created by the berms can then be equipped with separate effluent drain systems and appropriate influent/effluent valves. Separating a BCR into units provides opportunity for maintenance of a portion of the system (such as removal of substrate) while the other units continue to treat MIW. Building BCR units in parallel or sections is part of an overall design approach to improve capability and efficiency of maintenance on substrate materials over time. The basis for this approach is discussed further inSection 4.2.5 and maintenance approaches are discussed in Section 6.0.

5.2.2.5 Cover Installation

As determined by the design, a cover over the substrate layer may be beneficial. This determination is mostly based on climate (Section 2.6). Cold winter climates can have adverse effects to the BCR unit sulfate reduction rate due to the low temperature. Also, a cover may be desired based on the designers' preference and experience. Covers can also help with reduction in odor issues caused by hydrogen sulfide gas production from the sulfate reduction reaction.

The cover system can consist of any number of materials, preferably overburden or other on-site materials leftover from the BCR construction to reduce costs. Placement of cover material must also be carefully conducted to limit compaction to the substrate layer. As such, a higher density material such as gravel or rock may not be preferred. Layers of wood chips have been used at a few sites. The placement and grading of cover material should be installed from the perimeter of the system using excavators. If equipment must be driven onto the BCR unit surface, sufficient cover material should be installed first within the area that the equipment operates.

Cover material generally should be sufficiently sloped to shed stormwater off of the BCR unit surface, assuming the cover extends to the top of the BCR's containing berm. Grade checks should be conducted periodically to ensure the designed cover thickness and grades are met. The cover should also be graded and tied into runoff/run-on diversion structures along the perimeter of the BCR unit, if present. Routing water from the BCR unit surface towards the influent or effluent areas should be avoided. Otherwise, inflow of runoff water into these areas could increase the amount of water to be treated in the BCR or in subsequent posttreatment steps.

The design may consist of a low permeability cover system, which would involve a material similar to the BCR unit liner that limits infiltration. Placement of this kind of cover should be conducted similar to the methods discussed for liners in Section 5.2.2.2, depending on the type of material. The low permeability liner should have a protective layer placed over the liner and may also have a seeded surface. This layer protects the liner from photodegradation and heave/thaw affects that could damage the liner. Ventilation of gases (hydrogen sulfide) within the BCR unit may be a required component of the cover system. If this is required, vent pipes would be installed through the cover system. Penetrations in the liner for the vent pipes should be installed as described for liners installation in Section 5.2.2.3.

5.2.2.6 BCR Systems Effluent Control Installation

Treated effluent from a BCR can leave from a constant elevation overflow pipe via an open or valved piping system, or as an overflow pipe or overflow spillway from the BCR. There are also adjustable water head control valve boxes that allows a predetermined water head in the BCR unit to be maintained. Figures 5-5 and 5-6 present a constant head overflow discharge pipe and an adjustable head overflow device, respectively.



Figure 5-5. Example constant elevation BCR overflow discharge.



Figure 5-6. Adjustable elevation head weir for BCR effluent.

Mechanical failures from the effluent system include the freezing of the effluent control valve that controls the effluent for the BCR, if present. Although a good valve exercise program minimizes the potential for the valves to become stuck, the presence of solid mineral deposits including iron and other metal hydroxides or sulfides make the BCR effluent valves susceptible to failure. Also,

the adjustable weir boxes used in BCR effluent systems are also prone to become stuck due to the presence of mineral build up.

5.2.3 BCR Systems Performance and Monitoring Equipment Installation

The BCR units are constructed with some monitoring wells and piezometers that allow for monitoring the performance of the BCRs as well as the performance of the BCR liner. Figure 5-7 presents a BCR with monitoring wells and piezometers. In addition, there may be monitoring ports within the BCR itself (see Section 4.3.4). These are normally used to ensure that the hydraulics of the BCR are acceptable and that there is no apparent short circuiting of flow as well as providing a location where water samples can be collected within the BCR unit. These monitoring wells can provide information on the performance of the BCR, the location of the anoxic zone, and location where substrate may be compromised.



Figure 5-7. BCR monitoring wells and appurtenances.

In addition, sometimes it is necessary to cut and extract a vertical transect of the BCR to ensure that the various BCR zones are clearly defined in the cross section. This type of sampling is rare but can be extremely useful during the diagnosis of the BCR performance.

Most MIW BCRs do not contain automatic flow and sampling devices and rely on the manual monitoring performed by experienced personnel, including trained private companies, federal or state personnel as well as local organizations and site stakeholders representatives. The design of the BCRs normally provides sufficient technical information to determine a raw and treated effluent chemical characterization that the samplers are familiar with and compared results such as pH, temperature, ORP, DO, and actual water chemistry samples of heavy metals, sulfate, sulfide, alkalinity

> ***Watershed Groups and Treatment Systems***
>
> *Despite the presence of watershed groups able and willing to operate and maintain treatment systems in hard rock areas of the west, there are few funds to assist them. Presently, CWA Section 319 program funds often cannot be used at these western sites because many of the discharges are considered to be point-source discharges, thereby rendering them ineligible for 319 funding. See Section 8.0 for more detailed information.*

and other site-specific parameters, as designed (seeSection 6.2.2.2 for a comprehensive list of parameters).

BCR effluent monitoring commonly is comprised of a discharge pipe where effluent sampling using field monitoring equipment such as a stopwatch and bucket for flow determination, pH and ORP meters, and thermometer are used. Samples for off-site analysis can also be collected from the BCR effluent. Figure 5-8 shows a BCR discharge pipe with field equipment being used.



Figure 5-8. BCR effluent monitoring.

Automated monitoring equipment include open channel primary and secondary flow measuring devices as well as automatic samplers that can be programed to collect BCR effluent (and influent) samples based on a predetermined sampling protocol. Section 6.0 of this guide presents a detailed description of automatic flow monitoring and samplers used for MIW BCRs. The precautions necessary for the installation of primary flow monitoring devices is to provide access and infrastructure for the installation of primary flow monitoring devices, including an open channel and weir system as well as conditioning the site to locate the flow and sampling equipment in a safe, secured and reliable location.

In addition a small wood shed or prefabricated shed is normally useful to keep sampling equipment and consumables needed for flow monitoring and sampling purposes. Sampling equipment may be operated by the use of solar power and solar panels and photocells need to be installed such that direct sunlight is maximized. In addition, sampling devices can be powered by water wheel technology.

5.2.4 BCR Systems Site Construction Restoration

The BCR system design drawings and specifications normally include a section on the site construction restoration that includes the minimum restoration design criteria after the BCR system is built. Site restoration can call for re-establishing the site not occupied by the BCR system to its original state, matching existing conditions to the area surrounding the work site or partially restoring areas, allowing access roads, building, and specific features to remain.

As the site restoration takes place, field QC of restoration activities may require approval by stakeholders, such as the appropriate grass seeding mix needed as well as documentation of restoration activities and meeting site restoration conditions. After topsoil placement and grading is complete,

the surface is normally roughened (raked) prior to applying fertilizer and grass seed. The topsoil is loosely compacted to promote growth.

Following placement of the impermeable geo-membrane over the bioreactor, general backfill and topsoil are often placed and loosely compacted with a final grade normally 1-foot above the surrounding grade to allow for surface runoff. Final grade along the perimeter of the BCR Unit can gradually blend into the surrounding surface grade.

All remaining disturbed areas (such as backfilled trenches, monitoring well completions, stockpile areas, haul routes, and support construction areas) may need to be restored to conditions similar to or better than pre-construction conditions. This restoration may include the addition of topsoil, fertilizer, and seeding as required.

Restored areas are normally maintained, protected, and sometimes fenced to limit access to the BCR system site. All restoration activities are normally coordinated with the owner or owner's representatives to ensure that restored areas are completed to their satisfaction.

5.2.5 Site Security

During BCR system construction, all visitors to the work site are normally required to report to the site construction manager as soon as they arrive on site. The area immediately surrounding the BCR system construction work area is normally marked through the use of signs, barrier rope or tape, or fencing. BCR system security is designed to prevent exposure of unauthorized, unprotected individuals to the work areas.

Due to their generally remote locations, BCR system construction site security can range from nonexistent to chained-linked wire fence, depending on the location of the BCR system and its proximity to urban areas. More sophisticated security systems such as camera and motion detector-based surveillance systems are normally not necessary as part of the BCR system construction, but may be necessary if the BCR system is adjacent to a site that requires surveillance.

6.0 SYSTEM START-UP, OPERATION, AND MAINTENANCE

Bioreactors used to treat MIW are generally designed for minimum operational maintenance, especially when located in remote areas. For example installation in a remote area where maintenance is difficult or unlikely can result in failure to provide treatment (see Appendix B.14, Lady Leith mine case study).

However, some level of operational controls and maintenance are necessary to achieve the performance expectations of the bioreactor and treatment system as a whole.

This section describes the following monitoring, operation, and maintenance activities:

- system start-up (Section 6.1)
- operation and monitoring (Section 6.2)
- maintenance (Section 6.3)
- sampling protocol (Section 6.4)
- content of an operation and maintenance plan (Section 6.5)

The monitoring section describes monitoring tools and techniques for the physical condition of the bioreactor, the system flow rate and water chemistry for operational performance, and water chemistry for regulatory compliance. The maintenance subsection includes substrate/nutrient replacement, troubleshooting and preparation of an operation and maintenance plan.

6.1 System Start-up

Start-up should allow sufficient growth of the microbial populations to establish a stable biomass within the reactor unit. As described in Section 3.0, the source of the microbes often is from manure (Table 1-1), but may also come from any combination of laboratory designed inoculum, wastewater, anoxic site sediments, or source water (Gusek and Figueroa 2009; La et al. 2003; Perrault et al. 2009). Appropriate chemical and physical conditions are necessary to promote the growth of the desired microorganisms. The sulfate reducing bacteria require anoxic or very low dissolved oxygen levels, sulfate, and an organic carbon food source.

There are generally three phases of start-up:

1. Reactor flooding
2. Low-flow or recirculated flow operation
3. Full-flow operation with frequent monitoring

BCR Start-up

During a two week start-up, pumps recycled the water within the two BCR cells. Based on data collected in field, and subsequent laboratory confirmation, the water from the BCR cells was routed to the tailings pond for temporary storage and later treatment and release. At that point, the rock filter and aeration ponds were brought on-line. In the meantime, the mine discharged according to plan through an overflow pipe from the settling pond as it had during construction of the other components. See Section B.7 for more details.

To initiate the required conditions, system flooding allows the microbes to incubate. This flooding wets all the material in the treatment unit, and allows soluble carbon and sulfate to come into contact with the microbes thereby allowing establishment of attached microbial growth without washing food and microbes out of the reactor unit. During this flooded period microbes use oxygen in the influent water to consume organic carbon. These microbes may continue to be present in the system during normal operations and many aerobic microorganisms are facultative instead of obligate aerobic microorganisms.

Generally, flooding of the system is conducted by shutting off the effluent valves, allowing the influent water to fill the treatment units, and then bypassing any additional source water around the treatment unit for the duration of the flooded period. The second phase can include a recirculation period that is intended to circulate dissolved organic compounds and sulfate through the reactor to stimulate the growth of sulfate reducing microbes in all portions of the reactor. A two week flooded period followed by a four week recirculation period was successful for the start-up of the biochemical reactor used at the Standard Mine (Reisman et al. 2008). The second phase of start-up may instead include a recirculation period that mixes the biochemical reactor effluent with a small portion of untreated source water as influent to the treatment unit. During this type of recirculation phase, the majority of soluble organic compounds and dispersed microbes, that would otherwise be flushed out in a one-pass operation, are returned to the head of the reactor, while still providing a portion of fresh influent containing sulfate.

At sites where recirculation of water may be impractical (due to lack of power source for pumping), initially maintain very low flow rates of influent source water and allow the system to slowly adapt to increased flow and loading over a several month time period. This practice is presumed to allow the biological system to establish attached populations that might otherwise be washed out of the system if high flows are used immediately. Slowly increasing the flows, and corresponding loading, allows the biological system time to accumulate and adapt to conditions that provide effective treatment. Generally greater success and shortened time for start-up will be achieved with warmer weather and water temperatures, therefore, starting operations in the summer months is preferable to the winter months in cold climates. (Riesman 2011, personal communication).

The success of flooding and recirculation or low flow, and determination if additional time is needed in these phases, is evaluated by monitoring certain water chemistry parameters in the influent and effluent water and in sampling ports within the biochemical reactor unit. The four key parameters are:

- sulfate concentration
- dissolved oxygen (DO)
- oxidation-reduction potential (ORP)
- pH

These parameters should be measured on a weekly or more frequent basis, if possible, in the influent to the reactor unit, the effluent of the reactor unit, and sampling ports within the reactor unit, if available. The difference between source water and reactor unit sulfate concentrations will indicate if sulfate reduction is occurring. Lower ORP values in the biochemical reactor unit will indicate if oxygen has been removed and conditions are suitable for sulfate reduction. An increase in pH from source water may indicate the commencement of sulfate reduction, which produces alkalinity, or it may be due to other inorganic alkalinity sources provided in the reactor, but either way, a pH of 5 or greater (toward neutral) generally promotes more rapid growth and activity of sulfate reducers. Additionally, field assays for sulfide performed by an experienced field technician can be used to supplement or replace sulfate measurements to determine if sulfate reduction is occurring in the reactor.

Once sulfate reduction is occurring in the reactor as evidenced by the key water chemistry parameters, the system can be gradually brought online to receive full flow by incrementally rerouting the previously bypassed source water flow to the reactor, or where recirculation is used, by gradually increasing the ratio of source water to recirculation flow to eventually eliminate the recirculation. The initial full flow operation is the third phase of start-up operations. During this third phase frequent sampling of flow rate and chemistry of the influent, effluent and internal reactor sampling ports, if available, are recommended in order to understand and document the performance of the system including sulfate removal rates, metal removal efficiency, ORP, and pH conditions. The specific water chemistry parameters are discussed further in Section 6.3. Generally weekly sampling during this period is suggested if feasible. The full flow phase of start-up is an appropriate time for a tracer test of the system. A tracer test is not required, but establishes an observed residence time distribution at start-up for later comparison if short-circuiting or blockage is believed to be developing.

The effluent initially released by a BCR once full flow is established is likely to be high in suspended solids, biological oxygen demand (BOD), and nutrients. The cause of these high values is believed to be the flushing of fine solids and dissolved compounds from the substrate. The initially high values have been observed to decline rapidly to acceptable levels, generally requiring a few weeks of operation. Presumably this correlates to the flushing out of the soluble materials.

In addition to monitoring the biochemical reactor effluent during the third phase of start-up, it is also important to monitor and evaluate the effluent of all system components including the pre- and

posttreatment units such as aerobic polishing units or settling ponds. The point of compliance for the entire treatment system should also be monitored at more frequent intervals during this third phase of start-up.

As a caution: it is possible to leave the biochemical reactor in the flooded period too long. An extended period of flooding may generate a buildup of metabolic byproducts, including high levels of sulfide that have been found to be toxic to the bacteria (Luo et al.2008). Generally, the conditions can be returned to promote the growth of sulfate reducers by providing fresh influent source water and allowing waste products to exit the system in the effluent.

6.2 Monitoring and Maintenance Activities

Biochemical reactor systems generally have a lower operation and maintenance requirements than traditional active treatment plants. However, a certain level of operations, monitoring, and maintenance is required. The level of effort required depends on many factors including chemical and hydraulic loading rates, variability of loading rates, design, location, and the degree of flexibility in the discharge criteria.

This section describes the operations and monitoring that should be considered for a biochemical reactor. The main objectives of operations and monitoring are to manage the system for smooth operation and minimal upsets and to determine compliance with target water quality criteria. To achieve smooth operation it is important to maintain flow and loading within the designed system limits and to regularly assess system conditions and performance as a predictor for potential upsets, necessary maintenance, decline in effectiveness over time, and end of useable life.

Three main types of monitoring to consider are as follows:

1. monitoring the physical condition of the system
2. operational performance monitoring of system flow rates and water chemistry conditions
3. monitoring water chemistry for regulatory compliance

6.2.1 Monitoring the Physical Condition of the System

Monitoring the physical condition of the system generally includes visual inspection of the overall system and site and the various components of the system, including pre- and post- treatment units. Observe changes from normal operations and track changes that may occur slowly over time in order to identify and mitigate minor problems before they cause major system upsets. Regular inspection should include monitoring flow through the system, system integrity, and site access.

It is important to maintain flow through the biochemical reactor units and identify if preferential flow or clogging is causing areas to dry out or become stagnant. Either of these situations will impact treatment performance because of the loss of treatment area. The following should be included in monitoring system flow:

- Monitor hydraulic head with piezometers, gauges or standpipes or other designed feature at the influent and effluent to treatment units.
- Check operation of gravity feed, look for clogging and water back up.
- Inspect drop inlet/outlets and weirs, look for clogging and water back up.
- Ensure pumping systems, if used, are operational and that required pump maintenance is performed as required by manufacturer.
- Check for clogged piping, short-circuiting, and channelization. These issues may or may not be evident with simple visual inspection. Systems may be designed with features for back flushing or changing flow path, and these operations should be performed as suggested by the system designer but may be on the order of once to twice a year.
- Check flow through pre- and posttreatment components to the system.

Maintaining system integrity can help to ensure an operational system. Monitoring for system integrity should be performed as a walk through with visual inspection of the following features:

- Inspect dikes, berms and other containment or erosion control features for physical damage from stormwater events or animal disturbance.
- Inspect any substrate feed equipment including pumps and power sources.
- Inspect usability of any sample ports and automated instrumentation.
- Inspect for animal activity in and around treatment units including the presence of water loving or burrowing animals (such as beaver and muskrat). Insects and poisonous snakes may be problematic from a health and safety perspective.
- Monitor for hydrogen sulfide is important for health and safety, generally levels at an open air facility will not reach toxic concentrations but low lying areas or manholes could accumulate this toxic gas.
- Monitor for odors especially in areas that have nearby residents, since even a small amount of hydrogen sulfide odor may be unacceptable to neighbors.
- Inspect that vegetation occurs in and around biochemical reactor and treatment system only in areas designed to support vegetation:

    - In some cases vegetation may cause problems with flow paths and system chemistry, in other cases negligible impacts have been observed.
    - Some posttreatment processes may rely on vegetation for functional processes.
    - Excessive vegetation external to the bioreactor may impede access or tree roots may compromise subsurface liners or berms.
    - Invasive species can also become a maintenance issue depending on the land management practices in the area.

Maintaining and controlling site access is key to keeping systems operating smoothly by facilitating the ability to access the site when problems arise, as well as limiting potential problems caused by unauthorized persons. Monitoring site access should include the following:

- Inspect access road conditions.
- Inspect access restriction structures such as fences and locked gates.

- Post emergency numbers at the site so anyone witnessing an unusual condition (for instance, a berm break or leak, higher than normal odor, flooding, or vandalism) can contact site personnel.

The frequency of physical condition monitoring inspections largely depends on site accessibility but is suggested at once a month when feasible. If influent loading conditions or other site conditions vary, more frequent monitoring may be required.

6.2.2 Operational Performance Monitoring of Flow and Water Chemistry

Both influent and effluent flow rates and water chemistry monitoring are important to assess the operation of the biochemical reactor unit. Frequency, location, and water chemistry parameters suggested for monitoring are discussed in the following subsections. Table 6-1, Sample Locations, Frequency and Purpose of Sampling, summarizes the sampling locations, frequency and purpose of sampling.

6.2.2.1 Sampling Frequency

A monthly operational monitoring frequency has been established for several demonstration sites (Bless, et al. 2008). In some cases, monitoring is increased to once per week for field parameters and once per two weeks for laboratory analytical parameters in the summer months when remote sites are more accessible (see Section B.8). Always be cautious about sample frequency through changing seasonal conditions and the effect these seasonal changes can have on a sample or on a site's accessibility for collecting samples.

6.2.2.2 Water Chemistry and Quality Parameters

Water chemistry parameters generally consist of those that can be measured in the field with probes, meters or colorimetric tests kits and those that require laboratory analysis.

Field parameters, their importance and their general purpose are listed below under required and optional categories. Required field parameters include the following:

- Flow rate – used to track system loading, monitors if treatment flow differs from designed treatment flow, recording of any overflow or treatment by pass events
- Temperature – tracks changes in the system, very cold temperature can cause low microbial activity or poor settling of solids
- pH – tracks if pH is changing through the reactor units and is within the designed target range, usually between 5 and 8.5 but variations in design pH exist
- Oxidation reduction potential (ORP) – tracks changes in how oxidizing or reducing the water conditions are through the biochemical reactor unit. Low ORP indicates reducing conditions suitable for sulfate reduction. Microbes that use oxygen and other higher energy electron acceptors (i.e. nitrate) generate the low ORP conditions suitable for sulfate reduction. High ORP inside the BCR substrate or BCR effluent can indicate that microbial activity is

limited, higher levels of oxygen are entering the system than anticipated, or short circuiting is occurring.
- Alkalinity – can be measured in the field or in the laboratory, tracks changes through the system and reactor units that are due either to alkalinity production from sulfate reduction or alkalinity addition from chemical or physical substrate
- Visual observation of effluent color – black precipitate in the biochemical reactor effluent water indicates the presence of metal sulfides and therefore is a good indicator that sulfate reduction is occurring. White precipitate, which is often elemental sulfur, is frequently observed and also indicates that sulfate reduction is occurring. Generally, in a well-functioning system the black color will be dominant in the effluent, while the white precipitate may be evident surrounding the outfall area.

Optional field parameters are those that provide supplemental information on the system or can be used as a substitute for a required field parameter:

- Dissolved Oxygen (DO) – indicates the level of oxygen in the system, oxygen-depleted conditions (below 2 mg/L as measured by probes) in the BCR are required for successful sulfate reduction. High DO within the BCR or BCR effluent can indicate that microbial activity is limited, higher levels of oxygen are entering the system than anticipated, or short circuiting is occurring.
- Iron and iron speciation – can be measured with a field test kit. Iron speciation identifies whether reduced (Fe^{2+}) or oxidized (Fe^{3+}) iron predominates in the system. Reduced iron is indicative of a well-functioning system. Iron concentration indicates whether iron loading (load = concentration times flow rate) is above load design. Also in some cases iron concentration in the effluent has been used as an indicator for compliance with other metals concentrations (see Section B.8);
- Acidity – can be measured in the field or in a laboratory, generally tracks the opposite of alkalinity and provides similar information;
- Electrochemical potential (Eh) – is an ORP measurement adjusted to standard hydrogen potential rather than some other reference electrode; and
- Specific Conductivity – often measured to monitor for significant changes, however, this parameter is generally not considered critical.

Laboratory parameters, their importance and their general purpose are listed below under required and optional categories. Some of the following can be measured in the field using appropriate test kits.

- Metals of concern – can include aluminum, arsenic, cadmium, copper, chromium, lead, iron, manganese, selenium, and zinc or other metals that may be present and regulated at the site. Metals are measured in the influent and effluent to determine removal efficiencies and to determine regulatory compliance with state and/or federal water quality criteria. High levels of iron and aluminum can increase clogging potential and are often monitored closely. Often

only dissolved metals are measured unless total metals are regulated at the site.

- Sulfate – tracks the rate and level of sulfate reduction in the biochemical reactor. Can be an indicator of healthy sulfate reducers and can be used to determine if enough sulfide, stoichiometrically, is produced to precipitate the load of metals in the influent source water.
- Alkalinity – can be measured in the field or in the laboratory, tracks changes through the system and reactor units that are due either to alkalinity production from sulfate reduction or alkalinity addition from chemical or physical substrate
- Biological oxygen demand (BOD) – used as an indicator of the concentration of readily available organic carbon in the system water. This parameter is used to ensure sufficient food source and may also be regulated in the system effluent to protect receiving water bodies.
- Chemical oxygen demand (COD) – tracks the oxygen demand from chemical oxidants. COD may also be regulated in the system effluent to protect receiving water bodies.
- Total dissolved solids (TDS) – may be regulated to protect receiving water body. TDS is not treated for in biochemical reactors but some removal may occur.
- Total suspended solids (TSS) - may be regulated to protect receiving water body. TSS is not treated for in biochemical reactors but some removal may occur.
- Monitoring of substrate use - If appropriate sampling access points are installed during construction, pre-placed packets or cores of the substrate can be accessed to determine how much solid substrate remains in the system. Typically sampling for substrate use is done after five to ten year of BCR operation. Cores or packets can be observed visually for assessment of remaining key components of the BCR substrate (hay/straw, wood chips or limestone) with depth. Laboratory assessment can also be done by calculating loss on ignition and comparing that to measurements from the original substrate. Lime can be separated from organic matter by careful washing or utilizing gold pan separation techniques and then dried and weighed and compared with a similar volume of the original substrate.

6.2.2.3 Sampling Locations

The following locations are recommended for operational monitoring: (see Table 6-1)

- source water influent to the entire treatment system
- Influent to the biochemical reactor unit, if pretreatment of source water occurs
- Internal biochemical reactor unit sampling locations, if applicable
- Effluent from the biochemical reactor unit
- Effluent from pre- and posttreatment components, includes final system effluent
- Point of compliance (discussed in Section 6.2.3)

In systems with redundant biochemical reactors or other redundant treatment units, the sampling locations described above should also be included for the redundant units.

Table 6-1. Sample locations, frequency, and purpose of sampling

Sample Location	Parameters	Frequency	Purpose
Source water influ-	Laboratory Analysis:	Monthly, possibly higher	Baseline influent for determining

Table 6-1. Sample locations, frequency, and purpose of sampling

Sample Location	Parameters	Frequency	Purpose
ent to the entire treatment system	Metals of concern, pH, sulfate, Alkalinity, (TDS, TSS if regulated)	frequency	and maintaining operational system performance and may be required for regulatory compliance as comparison with treated water quality (% reduction)
	Required Field parameters	Monthly, every two weeks if accessible and high variability of influent	Operational monitoring for determining system performance
Influent to the biochemical reactor unit (if pretreatment present)	Laboratory Analysis: Metals of concern, pH, sulfate, Alkalinity	Weekly to Monthly during start-up then this can be phased out once system is operating as anticipated. May be used for trouble shooting	Operational monitoring for determining system performance
	Required Field parameters	Monthly, every two weeks if accessible and high variability of influent	Operational monitoring for determining system performance
Internal biochemical reactor unit sampling locations, if applicable	Required Field parameters	Weekly to Monthly during start-up then this can be phased out once system is operating as anticipated. May be used for trouble shooting.	Operational monitoring for determining system performance
Effluent from the biochemical reactor unit	Laboratory Analysis: Metals of concern, (Kirby and Cravotta (2005), net acidity / alkalinity, sulfate, COD or BOD	Weekly to Monthly during start-up then this can be phased out once system is operating as anticipated. May be used for trouble shooting	Operational monitoring for determining system performance
	Required Field parameters	Monthly, every two weeks if accessible and high variability of influent	Operational monitoring for determining system performance
Effluent from pre- and posttreatment components, if not already captured, includes final system effluent	Laboratory Analysis: Metals of concern, net acidity/alkalinity, sulfate	Weekly to Monthly during start-up then this can be phased out once system is operating as anticipated. May be used for trouble shooting	Operational monitoring for determining system performance
	Required Field parameters	Monthly, every two weeks if accessible and high variability of influent	Operational monitoring for determining system performance

6.2.3 Compliance Monitoring

How treatment discharge is regulated can vary widely depending on the regulatory agency, geographical location, and receiving water body, as well as if the site is an abandoned or active mine site. Generally it is assumed that biochemical reactor systems will discharge effluent to surface water rather than groundwater. Surface water discharges to streams, wetlands and oceans are typically regulated by the Clean Water Act of 1972 (CWA) or State or tribe equivalent. In most cases active mine sites must comply with provisions of the Surface Mining Control and Reclamation Act (SMCRA) for coal mining and hard rock mining and secure permits for discharges to surface waters under the National Pollutant Discharge Elimination System (NPDES). Allowable contaminant concentrations regulated by NPDES permits are determined by the US Environmental Protection Agency's technology-based standards or toxicity-based standards established for stream uses, or the state or tribal equivalent. The permits are generally site-specific and may include total maximum daily loads (TMDL). Abandoned mine sites fall under other regulatory purviews including CERCLA, RCRA, SMCRA, brownfields, state, local, or tribal programs.

Flexibility in the location of the point of compliance has benefits. It may be best to locate this point at the point of discharge or perhaps below an acceptable mixing zone. All locations of the point of compliance should include protection of ecological receptors in the stream (for instance, anadromous or short freshwater reach to a marine outlet). Regulatory flexibility depends on the purpose of a treatment system, including whether the system is intended as an interim water treatment measure, as the long-term treatment solution, or whether the system is intended to provide just some level of treatment in an abandoned remote location that is otherwise impractical to employ more active water treatment systems (see ITRC MW-1 2010 Regulatory Issues http://www.itrcweb.org/miningwaste-guidance/reg_issues.htm). Often when BCRs are used as an interim water treatment measure while a longer-term plan is being formulated, certain water quality parameters may be waived that would otherwise be applied to treatment plant discharges. Often only pH and metals are targeted for regulatory compliance in these interim or very remote situations (see Appendix B.8, Leviathan Mine Case Study). Depending on the setting and goals of the system it may be possible to target percent contaminant removal for the system. However, when a bioreactor system is planned as the long-term solution in an accessible area, water quality parameters targeted by the regulator may be more comprehensive and may strive to meet tribal, state, or local water quality criteria or maximum contaminant levels (MCLs). A more detailed discussion on these issues is presented in Section 7.0.

It is important to determine what the regulatory point of compliance will be during the design of the system, as mentioned in Section 4.0. The entire treatment system that includes the biochemical reactor must be considered in determining the point of compliance (POC). The system may include naturally occurring areas that allow aeration and/or settling and if these areas are included as a functional portion of the system, they should be considered in POC selection. In some locations it may be unacceptable to use a natural area (i.e. natural wetland) as a functional component of the system. This must be determined in the design phase as it will impact where the POC must be and the design of the system to achieve compliance at that POC.

Frequency of compliance monitoring also can vary based on location, goals of treatment, and variability of influent flows or concentrations. The intended purpose of the system and the location of the system often dictate the required monitoring frequency. Remote mountainous locations may only allow infrequent site visits, particularly in snowy climates. It is anticipated that these sites also have more flexibility in the discharge water quality criteria as site visits may be practically limited to once a month or less due to accessibility (see Section B.8). Other sites that are implemented in accessible areas that discharge into sensitive water bodies can require more frequent compliance monitoring. The monitoring frequency should be agreed upon with the regulator in advance of system implementation and should be compatible with the goals of the system. If the goals of the system are to solely reduce contaminant loads in a remote location, then compliance monitoring frequency can be on the order of once a month or less. In contrast, a system that is expected to protect a critical water body should have a more frequent compliance monitoring frequency, once per week or once per two weeks depending on influent variability and anticipated system performance. Often if water quality criteria are based on 30 day average concentrations, a once per month compliance sampling event has been considered sufficient to determine compliance, particularly at remote or limited access sites (see Section B.8). Most importantly since these systems are often used in remote areas, a realistic compliance monitoring frequency should be established.

In some cases monitoring the influent is required, along with POC monitoring. Sampling the influent is also beneficial for determining percent removal for metals and sulfate. Water chemistry parameters for compliance monitoring generally always include pH and the dissolved fraction metals of concern. If selenium is a metal of concern, often the total recoverable concentration, rather than just the dissolved fraction, is regulated due to its mobility and ecotoxicity as a solid. The compliance monitoring parameters may differ from the operational performance monitoring parameters discussed in Section 6.2.2. Typically ORP, electrochemical potential, acidity, and alkalinity are not regulated, although these parameters are important indicators of the status of the biochemical reactor. Effluent levels of sulfate, BOD, COD, temperature, and total dissolved solids (TDS) and total suspended solids (TSS) have been regulated for certain systems even though some of these parameters are not targeted for removal by the biochemical reactor. Again, regulated water chemistry parameters are dependent on the regulatory agency, the receiving water body, and the agreed upon goals of the system. For example Michigan has narrative water quality standards for TSS (Rule 50 of the Michigan Water Quality Standards (Part 4 of Act 45) (http://www.michigan.gov-/documents/deq/wb-npdes-TotalSuspendedSolids_247238_7.pdf).

6.3 Maintenance

6.3.1 Substrate Nutrient Change Out

The life cycle of the BCR is dependent largely on loading of metals and electron acceptors and can vary greatly site to site. The following steps can be taken to assist in predicting when the solid organic substrate needs to be replaced or refreshed. The viability of the organic substrate can be monitored via the ORP front, which is the location in the bioreactor where conditions switch from oxidizing to reducing (reducing is desired). To determine the position of the ORP front, the bioreactor should be sampled horizontally and/or vertically and the samples should be analyzed for

metals, organic material and limestone. Several methods exist to determine this ORP front's location. If samplers of various depth (Figure 4-2) were designed into the BCR, these may be used with an ORP probe. A perhaps more reliable way to sample the vertical profile is to use ferrous sulfide gel (or ferrous oxide gel) inserted into a slotted pipe (similar to the sampler shown in Figure 4-2, but having perforations or slots along the whole length). The pipe, or a few such pipes, can in turn be inserted into the bioreactor, and left for 24-72 hours. The point at which Fe^{+2} turns to Fe^{+3} is the location of the ORP front.

To estimate the time frame for substrate change-out, the substrate should be sampled at 6 months and at 1 year after steady-state operation has commenced. The rate of progression of the ORP front can then be estimated. It is recommended that the bioreactor be resampled at the estimated half-life of the substrate to evaluate the reaction progress, then again near the time that change-out has been estimated to be needed.

The design of the BCR system may include liquid feed or solid substrate and may include pre- or post-treatment units where precipitated metals and excess biomass collects. All of these treatment units will require sludge or substrate removal at some point in the life of the system. Sludge removal efforts can be significant and need to be incorporated into the maintenance and funding plan. Sludge and substrate removal considerations include protecting treatment unit liners and piping, access and space for pumps or manual excavation equipment and dewatering equipment as well as transportation of solids to an appropriate off-site or on-site disposal facility. An example of sludge removal and dewatering from a liquid fed system is included in the Leviathan Mine Case Study (B.8). Sludge removal from collection ponds is not likely to be conducted frequently and therefore the periodic nature of this maintenance activity can be a challenge to managing funding cycles.

The waste substrate should be tested for toxicity to determine the appropriate disposal option. The Toxicity Characteristic Leaching Procedure (TCLP) test for hazardous characteristic (soluble metals) should be included as one of the tests. Additional testing would depend on state and local regulatory requirements, for example the state of California requires additional testing procedures and additional constituents beyond the TCLP.

To address short term changes in acidity of the bioreactor that may adversely affect the microbiologic population, limestone amendments can be added. Similarly, the organic substrate life may be extended by dressing the top with fresh organic material. A garden tiller (Thomas and Romanek 2002) or possibly a low-pressure tracked vehicle with a tiller may be used to mix the fresh organic into the substrate bed.

6.3.2 Troubleshooting

Two main categories of troubleshooting exist: chemical trends and physical trends. Chemical trends involve: 1) increasing trends in the concentrations of Fe, Al, and Mn in the influent, 2) decrease of reduced sulfur/increase in sulfate in the effluent, 3) increase in BOD/COD in the system effluent, and 4) change in Eh, pH in the reactor, and DO. Physical trends include plugging,

loss of effluent, overflow of bioreactor or influent pond, short circuiting, and loss of structural integrity of berms, dikes, ponds, erosion, and storm water control features.

Triggers are system operational limits (usually parameters measured in the system effluent) which, if exceeded, identify the need to take corrective action (response) to return the system to optimal performance. Examples of triggers include changes in metals concentrations, BOD, alkalinity and sulfate. The specific selection and numerical limits for triggers are unique to each bioreactor system. Triggers may be decided upon during system start-up, and should be agreed to by all parties involved in the operation and regulatory oversight of the bioreactor system. Triggers make the process of identifying and resolving problems within the bioreactor system much more efficient.

The following sections describe the parameters that should be monitored to understand problems or the potential for problems in a bioreactor unit. Where possible, triggers and likely response to the identified problem are included.

6.3.2.1 Field Data

This section provides "real time" guidance on BCR or system observations that may suggest potential performance problems that would be validated when results from analytical sampling are received (see Section 6.3.2.2). Each system is unique; the following discussion is focused on systems that are past the commissioning/start-up period and would be considered in "steady state" operation.

Note: some of the guidance provided may not apply in every situation; alternative configurations (such as upflow or horizontal flow) may create circumstances that are unique to a given site or system. The guidance is typically applicable for downflow/top-fed configured BCRs unless otherwise noted.

Water level (influent and effluent)
Water levels are easy to measure using staff gauges, staining on the sides of channels or BCR standing water surfaces. For a top-fed BCR, for example, there should be very little difference in elevation between the standing water on the surface of the cell and the water level in the discharge piping (see Figures 5.5 and 5.6). If there is, this might suggest blockage in the effluent piping, loss of permeability in the substrate, or other blockage-related conditions that might cause the water level above the substrate to rise. A BCR exhibiting a relatively large (greater than 25% of the substrate thickness) head loss through the substrate should trigger an investigation into why it is occurring.

Sometimes, there may be evidence of differential water levels that no longer exist. For example, staining or flotsam on the shoreline of a BCR could suggest that the standing water level could have been higher than a level currently being observed during an inspection. Inspectors should be vigilant for other observations that might provide clues to potential future problems.

If portions of the substrate are exposed above the typical standing water level in a BCR, the water levels within these exposed substrate zones should be compared with open water. A laser-leveling

instrument might be required to do this, since minor differences can indicate a short-circuiting condition.

Flow Rate
Most BCR treatment systems should have the ability for measuring flow rate. Unless a BCR is being closely monitored for evaporation losses, it is unlikely that flow data would be collected on the influent and effluent. Whether the flow rate is visually estimated from a staff gauge in association with a weir or flume or data is electronically collected on a data logger, any site inspection should include a secondary/validation flow measurement using "fool proof" flow measuring techniques such as bucket and stop watch or other simple methods. Differences in expected flows versus observed flows during an inspection may suggest a leaking BCR liner or the unseen effects of vandalism.

Close attention should be paid to stains and/or debris on staff gauges for clues of unusual flow events. For example, an old trick might be to "dust" a removable staff gauge/dipstick in the protected stilling well of a flume or weir. If the water level rises in the stilling well in response to a high flow event, the dust would be rinsed off to the high flow mark. While the inspector would not know the exact duration or time of the high flow event, this observation would be a clue to check the climate data or values on a data logger for more details. Any evidence of excess flow above the design value (within expected limits) should trigger a focused evaluation of the analytical data for lingering aftereffects.

Evidence of Overflow
Some BCR systems have built-in safeguards to prevent overloading in high flow situations. These design features might include compound weirs or overflow devices. For troubleshooting, include inspections of points in the piping system, including any emergency spillway in the crest of a BCR, for evidence of excess flow patterns. This evidence may include:

- ponded water downstream from a design overflow/bypass weir
- staining on pipes, tanks, and exposed geomembrane liners
- flotsam adhering to BCR "shoreline"
- unexpected high readings in flow depth data downloaded from data loggers

Overflow may be related to a number of abnormal operating conditions. Every BCR installation will have its own set of overflow related consequences. Thoroughly describe these consequences in any operation and maintenance manual that is developed for the system.

Effluent Odor
Despite the unpleasantness, the rotten-egg smell of a BCR that is reducing sulfate is a reliable indication that it is functioning well. Care should be taken in inspecting portions of the BCR where hydrogen sulfide (H_2S) gas could accumulate; i.e., the effluent piping and any subsurface structure where BCR effluent piping could vent.

Regardless, a faint presence of H_2S in the effluent is a sure sign that a BCR designed for sulfate reduction (as opposed to nitrate or selenium reduction) is probably working well and subsequent analytical data is unlikely to be disappointing.

Oxidation Reduction Potential (ORP)
Oxidation reduction potential readings are one of the easiest ways to evaluate the relative health of a BCR quantitatively. When troubleshooting a BCR, it is recommended that the ORP of the BCR effluent be measured first, when the inspector arrives on site. The reason for this is that the oxidizing conditions of very acidic MIW (or the standardizing solution) can "poison" the platinum electrode and cause it to produce readings that are not representative of true BCR effluent ORP. To demonstrate whether this effect can bias the ORP at a given site, inspectors/samplers should measure the ORP of the BCR effluent, then the influent and repeat the effluent measurement to see if the initial reading is replicated.

ORP readings of zero millivolts or below are reasonable indications that the BCR is functioning as designed. Readings above this threshold might suggest current or recent overloading.

Dissolved Oxygen (DO)
If an ORP meter/electrode is not available, a DO reading in the BCR effluent may be a reasonable surrogate field observation. The BCR effluent should exhibit DO readings of 1 mg/L or less. For comparison, fully oxygenated water should have a DO of 8 mg/L or more. If the BCR effluent is too oxygenated based on this criteria, recent or current overloading or substrate organic matter depletion might be suspected.

pH
BCR effluent pH should be circumneutral. If it is less than 6.0 standard units, recent or current overloading should be suspected. If the pH of the influent is greater than the effluent, fermentation bacteria (which typically cause pH decreases) may be too populous. The short term remedy for this situation might be to add limestone of some other form of buffering alkalinity to the upper foot of the substrate (top-fed cell). Revisiting the bench and pilot data upon which the substrate design was based may be needed to see if this phenomenon was previously observed.

Conductivity
BCR effluent conductivity should be similar to the influent values under steady state conditions. If the effluent conductivity is greater than the influent, this suggests that limestone in the substrate is being consumed. This might be observed during start-up but as the microbial community in the substrate matures, the reduction of sulfate and the concomitant production of alkalinity (see Section 1, Equation 11 and A.1.1) should reduce the rate of limestone dissolution and the influent and effluent conductivity values should approach a similar value. If the effluent conductivity is higher than the influent after a year's operation, overloading should be suspected. Again, bench and pilot data might be revisited to assess whether observations during steady state operation have a precedent unique to the site/system.

Sulfate ion contributes to conductivity. If the effluent conductivity is significantly less than the influent, this may be the result of significant biological activity, especially during warmer weather when the influent MIW temperature might be relatively elevated.

Visual Observation in Effluent

A white precipitate on rocks or plants near the BCR effluent is another indicator that a BCR may still be reducing sulfate to sulfide. This material may be native sulfur; another bacterial community will probably be present to oxidize it back to sulfate. This should be verified through laboratory analysis. The precipitate may be a variety of other mineral that indicate the BCR may not be working as expected. This micro-ecological relationship is one of the reasons that H_2S odors in the BCR effluent are not overpowering. The absence of this precipitate may be due to a relatively low sulfate concentration in the influent MIW or a well-matched rate of metal loading and sulfate reduction bio-kinetics. However, there is typically excess sulfate present in MIW and excess sulfide in the BCR effluent as well. Figure 6-1 illustrates typical visual evidence.



Figure 6-1. Biosolids cloudiness visual evidence of BOD at the Peerless Jenny King BCR in Montana, August 2007.

Source: D. Reisman

BCR effluent may be clear immediately after sample collection in steady-state conditions. However, if the sample is allowed to stand for a few minutes, it should become cloudy with suspended biosolids as the sample equilibrates with the oxygen in the air. This visual observation is a surrogate test for biological oxygen demand. The cloudiness indicates acceptable BCR operation as the bacterial community degrades the organic fraction of the substrate. During start-up, these suspended biosolids may clog sampling filters, so the samplers should be prepared with high capacity filters or be satisfied submitting unfiltered samples for laboratory analysis. In short, the presence of cloudiness in the BCR effluent that is allowed to stand is an indication that the unit is functioning well.

Abnormal climatic events (precipitation)

An abnormal weather event at a BCR site should trigger a site inspection just to confirm that the

BCR unit survived. High spring runoff flow falls into this category. For remote sites, it may be difficult to know when a major weather event occurs. From a troubleshooting perspective, if an abnormal weather event is suspected, inspectors should be vigilant for evidence that the BCR only sustained direct precipitation that fell on it and did not capture runoff from other parts of the site.

For example, a Forest Service road traverses the mountainside above the *Peerless Jenny King* BCR site in Montana (see Section B.10). An inspection the spring that followed construction revealed a fresh deposit of alluvial sand and debris in one corner of the upper wetland zone which was intended to polish iron up-gradient of four BCR cells. Expanding the inspection to the adjacent areas revealed an abandoned cabin filled with sand up to the windows. Eventually, the source of the sand and debris was traced to the Forest Service road where a roadside berm had been breached, probably in the seasons prior to the BCR construction. The abandoned cabin functioned (at least temporarily) as a sedimentation basin.

Troubleshooters should be prepared with rudimentary hand tools to manually correct issues such as the one described above and/or fix damage that could affect system performance.

Pipeline Scaling

Many MIWs contain dissolved and suspended iron and this situation is one of the largest nuisances for influent piping to BCRs as well as other water treatment systems. Iron oxy-hydroxide scaling can affect the accuracy of flow measuring devices (e.g. weirs and flumes) and/or restrict flow to the extent that the BCR only receives a fraction of the source MIW or it is completely by-passed (see Section B.14). While MIW delivery pipeline scaling will be a continuous maintenance headache, a few trouble-shooting tips that may be helpful (assuming that pipeline maintenance was included in the system design) follow.

Collect and analyze MIW close to or at the source and (especially if a long delivery pipeline is included in the design) and do the same near the BCR inlet structure. Comparing the iron concentrations over time should suggest that iron scaling is accumulating in the pipelines. If the MIW is not well buffered, one might even detect pH decreases as the dissolved iron hydrolyzes and hydrogen ions are produced.

Most designers know that iron scaling occurs more frequently at valves, fittings, flow restrictions, changes inflow regime (progressing from open channel flow in a pipe to full pipe flow in a "dip" in the pipeline profile), and drainage collection galleries.

One temporary and inexpensive scaling remedy that may buy some time (until a pipeline cleaning unit can be mobilized) is adding massive doses of sugar to the source MIW before it enters the scaled zone in the pipe conveyance (Gusek 2012). As the TOC in the sugar reduces the ORP of the MIW, the ferric iron in the iron oxy-hydroxide scale is reduced to more soluble ferrous iron and the scale dissolves. This technique was successfully used at a BCR site in Arizona.

Caution: while this sugar remedy may temporarily fix the physical scaling problem, the dissolution of iron oxy-hydroxide is likely to mobilize other MIW chemical parameters that may be adsorbed to the scale such as arsenic (if it were originally present in the MIW even in trace amounts) and

spikes in these parameters should be expected in the BCR influent. It is recommended that scale samples be analyzed prior to implementing this remedy to ascertain what might be expected.

Settling
Long-term settling of substrate as it degrades has yet to be documented. When vegetation establishes on BCR surfaces (whether by design or on a volunteer basis) and the roots and detritus from the plants accumulates, the steady addition of organic matter can mask settlement. Wildeman, et al. (1993) cited "…that biomass accumulation rates in tropical primeval coal swamps approached one foot every 10 years (Bateman 1950)."

Short term "settling" of substrate was observed during the construction of the Fran Mine pilot scale BCR in Clinton County, PA (see Section B.5). In this instance, the surcharge load of a soil cover being placed on the substrate induced compaction, which is not the same effect as settling. As the construction contractor noted the apparent settlement, additional soil was placed to remedy the situation which caused even more compaction. The final cover lacked a crown and in fact exhibited an obvious depression in the center of the cell. This effect should be anticipated in design of BCR covers; and compaction effects on BCR substrate (whose mixture proportions could vary due to field changes) should be verified in the field prior to cover material (i.e., soil) placement.

6.3.2.2 Chemical Trends

Iron
High concentrations of iron in the influent water may form ferric oxyhydroxides that may clog the pore spaces in the bioreactor (Section 4.2.1.1 and Appendix A.1.2). To avoid formation of ferric oxyhydroxides, keep iron levels in the influent below 100 mg/L at pH above 4.0. If the influent water is reducing, the iron concentrations could be higher, because the iron will remain dissolved under these conditions.

Aluminum
Aluminum can cause problems in some passive treatment systems by forming minerals that can cause plugging of the void spaces within the limestone in the system (Section 4.2.1.2 and Appendix A.1.3). In limestone-dominated passive treatment systems, the mineral gibbsite [$Al(OH_3)$], a gelatinous solid, may form. However, in BCRs, gibbsite does not appear to form and the unidentified aluminum-containing minerals that do form are less prone to cause plugging (Gusek and Wildeman 2002b). This was confirmed by Thomas and Romanek 2002, who identified non-gelatinous aluminum hydroxy-sulfate precipitates that formed instead of gibbsite in their limestone buffered organic substrate (LBOS)-filled BCR test columns.

Manganese
Manganese removal is generally poor in bioreactors (see Section 1.5.3, Equations 9 and 10, and Appendix A.1.4); however, reducing conditions typically found in bioreactors may release manganese bound in substrate at start-up. This situation is usually temporary until mobile manganese compounds have been depleted. If data show that manganese is being removed in a mature bioreactor, this may be a sign of overload, with manganese likely replacing calcium in the limestone, creating the mineral rhodochrosite ($MnCO_3$) (Gusek and Schueck 2004).

Sulfate
Under normal operating conditions in a bioreactor, sulfate ions are reduced to sulfide (see Section 1.3.3.4, Section 2.4, and Appendix A.2.1) by microorganisms and the sulfide reacts with dissolved metal cations to produce insoluble sulfides. Thus, one would expect to see lower sulfate concentrations in the effluent than in the influent. Monitoring sulfate concentration in the effluent therefore provides a measure of the bioreactor performance. If sulfate concentrations in the effluent increase with time, this likely indicates decreased metabolic activity of the microorganisms and decreased bioreactor performance.

Biological Oxygen Demand/Chemical Oxygen Demand
BOD and chemical oxygen demand (COD) are measures of the amount of oxygen potentially required to oxidize by organic chemicals present in the water. BOD measures the oxygen demand of biodegradable organic chemicals, whereas COD measures the oxygen demand of both biodegradable and nonbiodegradable organic chemicals. In a mature bioreactor, the effluent will ordinarily contain a steady state level of BOD. In the inlet piping and bioreactor cell, BOD should be totally dissolved. However, once the BOD-containing effluent contacts air, biological processes may cause the formation of suspended solids, which can cause pipe scaling. Effluent piping may need to be cleaned periodically to remove scale. The scale should be characterized to determine whether or not metals are present in BOD solids. If metals are present, effluent may be redirected back through the BCR.

Alkalinity
Alkalinity typically increases during operation of a bioreactor and approaches a steady-state level (see Section 1.3.3 and Appendix A.1.1). The increase is due to dissolution of limestone or as a byproduct of sulfate reducing bacteria. The increase in alkalinity buffers the acidic influent and helps to maintain the minimum pH of 4 - 6 needed for the sulfate-reducing bacteria to function. Sometimes the conditions within the bioreactor are sufficient for maintaining the optimum pH, but for extremely acidic influent streams the pH would need to be adjusted prior to entering the bioreactor (Gusek and Schueck 2004).

Nitrate
The presence of nitrate in the BCR influent can interfere with sulfate reduction as nitrate must be reduced before sulfate. If sulfate removal rates decrease or are poor for no other obvious reason, testing for nitrate may be warranted. This parameter could be inadvertently affected by seemingly unrelated site activity such as fertilizer application associated with revegetation work. If nitrate is already present in the BCR influent, it should decrease in concert with sulfate.

Mercury
The presence of mercury in the BCR influent should raise health and safety issue flags, as methyl mercury, which might form due to the activity of sulfate reducing bacteria in some circumstances, is extremely toxic. Methyl mercury can be adsorbed through the skin. If mercury is suspected in the MIW, a mercury speciation laboratory study using the proposed substrate should be mandatory. If excessive methylation is confirmed, the BCR technology may not be appropriate. It is suspected that mercury should preferentially form insoluble HgS in a BCR; mercury speciation testing in a

full scale BCR is strongly recommended.

Radionuclides
The presence of radionuclides (for example, uranium 238, uranium 234, thorium 230, radium 226 and others, Hofmann et al. 2000) in the BCR influent may trigger special handling of the BCR substrate during periodic retrofits that involve substrate replacement. The expected sequestration and accumulation of radionuclides in the substrate under reducing geochemical conditions may require a license and may trigger air emission (radon daughter) management. See the Nuclear Regulatory Commission website for more information.

Ca/Mg
Typical BCR substrates will contain limestone ($CaCO_3$), which may contain small amounts of dolomite ($CaMg(CO_3)_2$. The ratio of these two parameters in the BCR effluent compared to the influent should provide data that allows estimating the rate of limestone dissolution. The ratio of calcium and magnesium of the limestone used in the BCR substrate is required for this exercise; this information is typically provided by limestone suppliers/vendors.

In one instance, the removal of magnesium in a BCR has been cited (Gusek 2000):

> Four bench-scale test cells were constructed at the Smolnik mine and operated for two months to provide baseline information to design a larger pilot scale treatment cell. Two of the 200 liter-sized cells treated up to 5 liters of ARD per day. Results showed the viability of sulfate-reducing bacteria and limestone dissolution/exchange to yield metal loading reductions up to 98 percent of the influent values and pH increases to about 5.7. Remarkably, magnesium was removed in all four of the test cells, probably resulting from the formation of iron/magnesium carbonate in reducing redox conditions.

Magnesium removal should be viewed with suspicion and may be indicative of marginal BCR overloading.

6.3.2.3 Physical trends

Plugging
Plugging occurs when the flow in the bioreactor is blocked due to buildup of solids within the pore spaces. Examples of plugging include blockage of pore spaces in the substrate by iron and aluminum-bearing precipitates, and blocking of pore spaces in the organic substrate by metal precipitates. Plugging can be mitigated by controlling iron and aluminum loading in the bioreactor influent (this would be verified in bench and pilot testing), and adding rocks to the organic substrate to create larger pore spaces.

Note that Thomas and Romanek (2002) observed on a microscopic basis that aluminum hydroxysulfate and metal sulfide precipitates typically filled the individual cells within the cellulosic framework of wood particles in BCR substrate. Filling of pore spaces between the substrate particles with aluminous precipitates might occur in overloading situations which can be mitigated through design changes, if required. See below.

Overloading and Flow Control

The biological population within a bioreactor may be resilient to small, short-lived changes, but may suffer a significant decrease in performance if exposed to significant increases in metals concentrations or a decrease in pH, which can occur if the flow rate into the bioreactor exceeds design limits. This is referred to as overloading the bioreactor cell (see Section 4.4.2.4: Overflow Structures and Loading Control). Therefore, it is important to maintain flow through the bioreactor within design limits. Flow control in the full-scale system design can be achieved by several methods, including those in Gusek 2002b and 2004 and those described in Section 4.4.2.4:

- properly sized holding/mixing ponds
- flow-sensitive diversion ditches that are engaged only in overloading conditions (see Section 4.4.3)
- underground mine workings that are configured as storage reservoirs

Additional methods to maintain adequate flow through the bioreactor include exercising flow control valves (quickly open and close at least three times to flush out any iron scale in valve seat zone in the valve), and periodic cleaning of iron hydroxide scale deposits from the piping system.

Short Circuiting

Bioreactors are typically designed as vertical flow systems with the influent flowing downward (or upward) through an organic substrate. The effluent is collected at the bottom (or top) of the cell. When designing and constructing the bioreactor, it is important to avoid creating preferential flow paths (Section 4.4.5) whereby the influent can short circuit and avoid passing through the reactive zone of the bioreactor. Short circuiting is more likely near edges due to variation in hydraulic conductivity and has been repeatedly observed in lab-scale BCRs. In one case, 68% of the flow was found to be short-circuited through a lab-scale vertical BCR after slightly more than one year of operation (Fitch et al. 2008). In larger BCRs, such edge effects will be less likely to have a significant impact. During operation, evidence of short circuiting may include excessive total (unfiltered) metal sulfide concentrations with co-incidental low dissolved (filtered) metal concentrations. Carefully designed tracer testing by professionals may be required to identify short circuit pathways or zones in the BCR substrate.

The best method to determine short-circuiting is to perform a tracer test. A conservative tracer such as sodium bromide is mixed into the influent as a pulse or step-change and the effluent concentration is compared to the influent over time. The resulting curve shows the time distribution for the tracer, and thus fluid element carrying the tracer. Assuming some flow model such as a pair of parallel plug-flow systems, this residence time distribution (RTD) curve may be de-convoluted to find the apparent short-circuiting behavior.

Gas Lock-up

Gas lock up occurs when gases generated by biochemical and chemical reactions within the bioreactor collect within the piping and other locations within the cell. The gas buildup can cause a decrease in organic substrate permeability and short circuiting. These gases include hydrogen sulfide from sulfide ion, and carbon dioxide from bicarbonate ion or dissolution of limestone. In small-scale bioreactors, such as bench or pilot scale cells, piping is typically made of rigid material such as PVC or high density polyethylene (HDPE). Gas lock up is generally not a problem in these small systems due to short pipe lengths and rigid piping materials. However, some larger-scale bioreactors have been constructed with flexible HDPE pipe that can bulge upward and form gas traps. This can be a problem particularly in the gravel-filled drainage layer at the bottom of a bioreactor cell. Use of more rigid pipe and installation of gas vent pipes in the bioreactor cell are also methods to prevent gas lock-up. (Gusek 2004).

Gas Lock-up

The West Fork Mine Case Study includes an example of gas lock-up. The sulfide gas lock problem was investigated in December 1996 by installing vent wells in the substrate and measuring the gas pressures. Observations indicated that the gas was a factor in apparent short-circuiting of the water passing through the cell. The layered geotextiles (geonet and geofabric), originally intended to promote horizontal flow, appeared to be trapping the sulfide gas beneath them and restricting vertical flow. The permeability of the substrate itself was for the most part unaffected. However, construction practices in the south BCR cell could have contributed to the situation. Here, a low ground-bearing bulldozer was used to place substrate in nominal 6-inch (15-cm) lifts. This could have created a layering effect that may have trapped gas as well.

Structural Integrity of Berms, Dikes, Ponds

Maintaining the integrity of berms, dikes, ponds and other physical features of the bioreactor is critical to bioreactor function. Beavers and burrowing animals such as muskrats can cause significant damage to bioreactor structures. Muskrats are more likely to burrow in locations that are steeply sloped and vegetated, so keeping the slope to a 3:1 ratio and minimizing vegetative growth will reduce the likelihood of muskrat damage. Protecting the outside of the berm or dike with stone riprap (keep stone diameter to 6” or less to minimize holes where muskrats could enter) or covering surfaces at risk for muskrat entry with 1” wire mesh hardware cloth, heavy-duty plastic or fiberglass netting will also minimize the opportunity for muskrats to burrow into the bioreactor (Link 2005).

Beavers are attracted to the sound of running water and this triggers their digging and damming reflexes (Gusek 2004). To minimize the risk of beaver activity, the bioreactor cell can be designed with internal spillways or water level control structures that minimize the noise from falling water.

Erosion/Storm Water Management

Flow of water into and within the bioreactor should be maintained within design limits to ensure optimal bioreactor performance. The performance of the bioreactor should be evaluated after major

storms and corrective actions may need to be performed after major storms to restore bioreactor performance.

6.4 Sampling Protocol

Based on monitoring parameters described in Section 6.2, this section describes methods and equipment used to collect samples, and methods for in situ and laboratory analysis. Always check tribal, state, and federal rules for required or preferred sample collection and analytical methods.

6.4.1 Sample Collection

6.4.1.1 Instrumentation

A wide range of devices are available to collect field samples and obtain and process information (such as obtaining curves, trends). Generally, sampling probes are used, which extract defined water volumes. Probes may connect to a dedicated meter, which in turn can be connected to a computer for data collection and semi-instantaneous statistical analysis; some probes may instead be connected to computers through simple analog-to-digital cards.

Figure 6-2 shows an automatic sampler, which allows collection of water samples at a given frequency in three ways:

1. Collection of grab samples. There are samplers with multiple compartments, where each container us used for one sample taken at defined periods of time The volume and frequency may be set, for example, 50 mL every 5 minutes, or 100 mL per hour, and so on. The grab samples can then be analyzed individually of may be mixed to obtain a composite sample, or the samples may be composited into a single vessel.
2. Collection of composite samples proportional to flow rate. Individual sample aliquots can be collected proportional to the flow rate as programmed into the sampling unit and composited in a single vessel. Data collected during sampling is stored in an internal memory and can be printed only as data or as processed information in graphical form, for instance, flow versus time.
3. Collection of preprogrammed incremental samples at predetermined times (such as day and night, according to temperature ranges, or regular intervals).



Figure 6-2. Automatic wastewater sampler.

Source: CPTS 2009.

In all cases sample point selection must consider introducing the probe to the middle of water level as it is transported through open channels or closed pipes. In order to avoid external influences, like dust, the closer the sample point is to the source or outlet, the better.

6.4.2 Manual Sampling

Manual sampling should be minimized if possible due to the potential for human errors. For instance, in a composite sample situation, where samples must be taken each hour during the day for years, it is impractical to have an operator working in this way. The best option is to install an automatic sampler as described above. The cases in which manual sampling may be acceptable are when either a grab sample is sufficient or the sampling cannot be automated, such as ORP front sampling described in Section 4.3.4.

6.4.3 Instrumentation for In Situ Analysis

Most commonly used instruments for in situ analysis can be classified into:

- Portable meters: These meters allow rapid analysis at the site where the sample is taken. Generally speaking, these devices have sufficient precision as required for standard parameters such as temperature, salinity, conductivity, pH, DO, turbidity, redox potential, and COD.
- Permanent meters: These meters are installed in a permanent station that can measure one or more parameters constantly. The station consists of a sampling site, flow meter, sensor parameters, control unit and data logging.

6.4.4 Chemistry Sample and Analysis

The following two tables list recommended analytical methods used for field analysis (Table 6-2) and laboratory analysis (Table 6-3). Detailed descriptions of these methods can be found in Standard Methods for the Examination of Water and Wastewater (AHPA 2012; http://www.standardmethods.org).

Table 6-2. Field parameters, method, and purpose

Parameter	Name and standard methods	Description
Physical parameters		
Temperature	Thermometric: Method 2550	Consists of the immersion of a certified and calibrated thermometer in a water sample to establish the temperature. This parameter must be determined in situ.
Conductivity	Conductivity cell: Method 2510 B	Conductivity is the ability of inorganic salts in solution (electrolyte) to conduct electric current. Practically pure water does not conduct current, while water with dissolved salts does. The positively and negatively charged ions allow current flow, with resistance being inversely proportional to the concentration of ions. The conductivity values are relative to a reference temperature of 25°C. This method uses a Wheatstone bridge and platinum electrodes to determine the value of the conductivity.
Oxidation Reduction Potential	Electrometric: Method 22580B	Electrometric measurements are made by potentiometric determination of electron activity (or intensity) with an inert indicator electrode and a suitable reference electrode.
Chemical parameters		
Dissolved Oxygen	Iodimetric methods: 4500-O B 2. Membrane electrode method: 4500-O G	1. The iodimetric test is the most precise and reliable titrimetric procedure for DO analysis. It is based on the addition of divalent manganese solution, followed by strong alkali, to the sample in a glass-stoppered bottle. DO rapidly oxidizes and equivalent amount of the dispersed divalent manganous hydroxide precipitate to hydroxides of higher valency states. In the presence of iodide ions in an acidic solution, the oxidized manganese reverts to the divalent state, with the liberation of iodine equivalent to the original DO content. The iodine is the titrated with a standard solution of thiosulfate. 2. Oxygen-sensitive membrane electrodes of the polarographic or galvanic type are composed of two solid metal electrodes in contact with supporting electrolyte separated from the test solution by a selective membrane. The basic difference between the galvanic and the polarographic systems is that in the former the electrode reaction is spontaneous (similar to that in a fuel cell), while in the latter an external source of applied voltage is needed to polarize the indicator electrode. Polyethylene and fluorocarbon membranes are used commonly because they are permeable to molecular oxygen and are relatively rugged.
Ferrous Iron	Colorimetric	1. Method 1 uses individual, premeasured reagent pillows containing 1,10 phenanthroline for color development and absorbance measurement on a Hach DR/2000 spectrophotometer. This method is easily applied in the field, does not require a standard curve, does not require preparation of any

Parameter	Name and standard meth-ods	Description
		reagents, is specific for ferrous iron, and has a linear range of 0 to 3 mg L"1. Analyses are conducted per the manufacturer's guidelines (Hach Company 1992). Cuvettes were rinsed with deionized water (0.1 N HC1), and a final time with deionized water between readings. Samples above the linear range should be quantitatively diluted and reanalyzed. 2. Method 2 includes the use of FerroZine in 50 mM Hepes free acid to make a FerroZine reagent. The samples are analyzed by adding 1 ml of reagent to 5 ml of sample and then adding 4 ml of deionized water. A stand-ard curve can be developed using a blank and standards containing 1, 2, 3, and 4 mg L"1 ferrous iron. Samples with concentrations greater than 4 mg L"1 should be reanalyzed after dilution using 2 ml of sample and 3 ml of deionized water prior to addition of the reagent and additional deionized water. http://www.dtic.mil/cgi-bin/GetTRDoc?AD=ADA36119.
Alkalinity	Titration method: 2320 B	Hydroxyl ions present in a sample as a result of dissociation or hydrolysis of solutes react with additions of standard acid. Alkalinity thus depends on the end-point pH used.
Sulfide	Ion-selective Electrode Method: 4500-S2- G	The potential of a silver/sulfide ion-selective electrode (ISE) is related to the sulfide ion activity. An alkaline antioxidant reagent (AAR) is added to samples and standards to inhibit oxidation of sulfide by oxygen and to provide a constant ionic strength and pH.
Physiochemical parameters		
pH	Electrometric (pH-meter) or potentiometric: Method 4500 H+	The basic principle of electrometric pH measurement is the determination of the activity of hydrogen ions by potentiometric measurement using a standard hydrogen electrode and reference electrode.
Turbidity	Nephelometric: Method 2130	This method is based on the comparison of the intensity of light scattered by a sample under defined conditions or in the intensity of light scattered by a reference-standard suspension under the same conditions.
Color	Spectrometric: Method 2120 C	The color of a filtered sample is expressed in terms that describe the sen-sation realized when viewing the sample. The hue (red, green yellow, or other color) is designated by the term “dominant wavelength”, the degree of “brightness” is luminance”, and the saturation is by “purity” (such as pale or pastel). These values are best determined from the light transmission char-acteristics of the filtered sample by means of a spectrophotometer.

Table 6-3. Laboratory parameters, analytical method, and purpose

Parameter	Name and Stand-ard Methods code	Description
Physical parameters		
Total dis-solved solids (TDS)	Method 2540 C	A homogeneous sample is filtered through a fiberglass filter of standard dimension, and the filtrate is evaporated to dryness at 180 ° C in a tared container. The weight increase in the container relative to the dry, empty container represents the total dissolved solids.

Parameter	Name and Standard Methods code	Description
Total suspended solids (TSS)	USEPA Methods 160.2 and Standard Method 2540 D	A homogeneous sample is filtered through a fiberglass filter of standard dimension and solids retained on the filter are dried to constant weight between 103° to 105° C. The weight increase of the filter represents the total suspended solids.
Chemical parameters		
Sulfur	Iodimetric: Method 4500- S^{2-} F	Iodine solution, dissolved in the presence of potassium iodide (KI) is added to a defined sample volume. Iodine reacts with sulfur and the amount of excess iodine is determined by a reaction with sodium thiosulfate ($Na_2S_2O_3$), using starch as an indicator. If the solution turns blue excess iodine is present, and, if it remains colorless iodine is not present. The amount of sulfur present in the solution is calculated based on the difference between total iodine added and the excess iodine is determined by titration with sodium thiosulfate.
Alkalinity	Titration method 2320 B	Hydroxyl ions present in a sample as a result of dissociation or hydrolysis of solutes react with additions of standard acid until a reference pH is reached.
Acidity	Titration method 2310 B	Hydrogen ions present in a sample as a result of dissociation or hydrolysis of solutes react with additions of standard alkali until a reference pH is reached.
COD	Closed reflux and colorimetric determination Method 5220 D	Most types of organic matter are oxidized by a boiling mixture of chromic and sulfuric acid. A sample is refluxed in strongly acid solution with a known excess of potassium dichromate. After digestion the excess of potassium dichromate is measured against standards at 600 nm with a spectrophotometer.
BOD	Respirometric method 5210 D	Respirometric methods provide direct measurement of the oxygen consumed by microorganisms from an air or oxygen-enriched environment in a closed vessel under conditions of constant temperature and agitation.
Total Metals	Total Metals See Part 3000	Defined as the concentration of metals determined on an unfiltered sample after vigorous digestion, or the sum of the concentrations of metals in both dissolved and suspended fractions. Various methods can be used to quantify metals, like atomic absorption, plasma emission spectroscopy, voltametry, X-ray fluorescence, and titrimetric methods.
Dissolved Metals	See Part 3000	Defined as those constituents (metals) of an unacidified sample that pass through a 0.45 µm membrane filter. As in total metals a variety of methods can be used depending on the metal.
Ammonia	Ammonia-Selective Electrode Method: Method 4500 NH3 D	The ammonia-selective electrode uses a hydrophobic gas-permeable membrane to separate the sample solution from an electrode internal solution of ammonium chloride. Dissolved ammonia (NH_3(aq) and NH_4^+) is converted to NH_3(aq) by raising pH to above 11 with a strong base. NH_3(aq) diffuses through the membrane and changes the internal solution pH that is sensed by a pH electrode.
Nitrite	Colorimetric method: 4500-NO2- B	Nitrite (NO_2^-) is determined through formation of a reddish purple azo dye produced at pH 2.0 to 2.5 by coupling diazotized sulfamide with N-(1-naphthyl)-ethylenediamine dihydrochloride (NED dihydrochloride). The applicable range of the method for spectrophotometric measurements is 10 to 100 µg NO_2^-N/L. The color system obeys Beer's law up to 180 µg N/L with

Parameter	Name and Standard Methods code	Description
		a 1-cm light path at 543 nm. Higher NO_2^- concentrations can be determined by diluting the sample.
Nitrate	Cadmium reduction method: 4500-NO3- E	NO_3^- is reduced almost quantitatively to nitrate (NO_2^-) in the presence of cadmium (Cd). This method uses commercially available Cd granulated treated with copper sulfate (CuS_4) and packed in a glass column. After the reduction sample is analyzed as a nitrite sample.
Cyanide	Colorimetric method: 4500 CN- E	CN^- in an alkaline distillate is converted to CNCl by reaction with chloramine-T at pH <8 without hydrolyzing CNO^-. Caution: CNCl is a toxic gas; avoid inhalation). After the reaction is complete, CNCl forms a red-blue color on addition of a pyridine-barbituric acid reagent. Maximum color absorbance in aqueous solution is between 575 and 582 nm. To obtain colors of comparable intensity, have the same salt content in sample and standards.
Mercury	Dithizone method: 3500-Hg C	Mercury ions react with dithizone solution on chloroform to form an orange color. The various shades of orange are measured in a spectrophotometer and unknown concentrations are calculated from a standard curve.
Methyl Mercury	USEPA Method 1630	The sample is distilled with reagents to form methylethyl mercury (MEM) from methyl mercury. The MEM is purged from the sample with nitrogen onto a graphitic carbon trap. The MEM is thermally desorbed into an inert gas that carries the released MEM first through a pyrolytic decomposition column and then into the cell of a cold-vapor atomic fluorescence spectrometer (CVAFS). The concentration of MEM determined at the CVAFS corresponds to the concentration of methyl mercury.
Physiochemical parameters		
pH	Electrometric (pH-meter) or potentiometric: Method 4500 H^+	The basic principle of electrometric pH measurement is the determination of the activity of hydrogen ions by potentiometric measurement using a standard hydrogen electrode and reference electrode.
Turbidity	Nephelometric: Method 2130	This method is based on the comparison of the intensity of light scattered by a sample under defined conditions or in the intensity of light scattered by a reference-standard suspension under the same conditions.

6.4.5 Sample transportation

Transport of sample containers must be made in insulated coolers containing ice or cooling material. It is worth mentioning that the use of foam will help in preventing ruptures. Containers should be kept cool, dark and in a vertical position. Upon arrival at the laboratory, a chain of custody form must be filled out by the analyst and samples should be checked according to the shipping list. Samples must be immediately stored at 4°C until analysis.

Maximum periods of storage and preservation methods prior to analysis are included in Table 6-3. In any case, it is recommended that the time interval between sample collection and analysis must be minimized.

Table 6-4. Sample collection details

Parameter	Container	Sample volume [mL]	Type of sample	Preservation method	Maximum storage time
pH	P, G	50	S	Analysis in situ	2 h
Tem-perature	P, G	--	S	Analysis in situ	Seconds
Con-ductivity	P, G	500	S, C	Cooled	48 h
Solids	P, G	200	S, C	Cooled	7 d
Bacteria	P, G	100	S	Cooled	24 h
BOD	P, G	1000	S	Cooled	48 h
COD	P, G	500	S, C	Add H_2SO_4 until pH 2 and cool	28 d
Mercury	Teflon	100	S, C	Add HNO_3 until pH 2 and cool	28 d
Dissolved metals	P(a), G(a)	500	S	Sample must be filtered in situ, add HNO_3 until pH 2, store and cool	6 m
Nitrates	P, G	100	S, C	Cooled	48 h
Nitrites	P, G	100	S, C	Add H_2SO_4 until pH 2 and cool	48 h
Cyanide	P, G	500	S, C	Add NaOH until pH 12 and cool	24 h
P: Plastic; P(a): Plastic container previously washed with 1:1 HNO_3 G: Glass; G(a): Glass container previously washed with 1:1 HNO_3 S: Simple; C: Composite h: hour; d: day; m: month					

6.4.6 Field QA/QC

The USEPA (2002) guidance for use in developing a Quality Assurance Project Plan (QAPP) when designing an environmental data collection program is commonly used and accepted.

6.4.7 Laboratory QA/QC

In order to have a trustworthy result, the laboratory should follow adequate quality assurance/quality control standards, such as those listed in *Laboratory Documentation Required For Data Evaluation* (USEPA 2001). Additional information can be found in Hibbert (2007).

The following principles are important to consider when making reliable analytical measurements:

- Analytical measurements should be made to satisfy an agreed requirement.
- Analytical measurements should be made using methods and equipment that have been tested to ensure they are fit for the purpose.
- Staff making analytical measurements should be both qualified and competent to undertake the task.
- There should be a regular independent assessment of the technical performance of a laboratory.
- Analytical measurements from one location should be consistent with those elsewhere.
- Field duplicates, blanks, inter laboratory checks are important for laboratory control

- Organizations making analytical measurements should use well defined quality control and quality assurance procedures.

6.4.8 Flow Measurement Methods

Flow meters used to characterize the quantity of water, expressed as volume or mass per unit of time, are based on the fundamental equations of hydraulics (continuity equation), which are valid in open or pressure conduits:

$$Q = v \cdot A$$

where:
Q = Volume (in units of volume/time, m^3/s)
v = Velocity of fluid (in units of length/time, m/s)
A = Area through which the fluid has the velocity (v) (m^2)

Flow is determined using devices or primary instruments that are calibrated hydraulic structures, installed at a driving point of the fluid. Measurements can be obtained by determining the physical variables at specific points in the structure.

It is common to use a secondary instrument to determine the actual flow passing through the point of measurement, that is, capture of the signal emitted by a primary instrument (sonic means, light, electronics) and convert that signal to hydraulic data (flow, velocity, pressure). Measuring instruments can be classified into two groups:

- direct reading without registration or indirect reading from fixed points
- digital recording or graphic

The most common method of measuring flows for pilot and full scale BCRs is in open channel or free surface flow. Flow freely exposed to atmospheric pressure, like canals can be measured from the hydraulic loads (head) across a defined structure (such as the Parshall flume, a venturi or orifice plate) and water velocity (as through a turnstile), or using thermal, magnetic, or sonic techniques.

Meters most commonly used for BCR influent, effluent and overflows in open channels are rectangular weirs, V weirs and Parshall flumes. These devices can measure flow rates of water containing a certain amount of suspended solids like those resulting from BCR treatment. (Appendix D)

6.5 Contents of an Operation and Maintenance Plan

An operation and maintenance plan (O&M Plan) should be sufficiently detailed such that a new owner/operator of the site can take over operations without encountering major gaps in understanding the bioreactor system function. The O & M Plan should include a narrative report that describes the following:

1. The site background, location, and contact information
2. A description of the treatment system, including schematic diagrams
3. A description of flow measurement and water sampling locations and methods
4. A description of routine and major maintenance activities

At least one schematic diagram of the treatment system should be provided, and more than one diagram is recommended to show detail. One of the schematics should be an aerial-view diagram (preferred scale: between 1 inch:50 feet and 1 inch:100 feet). The schematic(s) should include a legend and should clearly depict the following:

- access roads
- each major system component, labeled
- normal flow path of water through system
- flow path of water during flushing operations (if applicable)
- valves or control structures used in regular Operation & Maintenance activities
- flow-measurement points, labeled
- sampling points, labeled
- emergency spillways and other flow control structures

An example schematic diagram is shown in Figure 6-3.

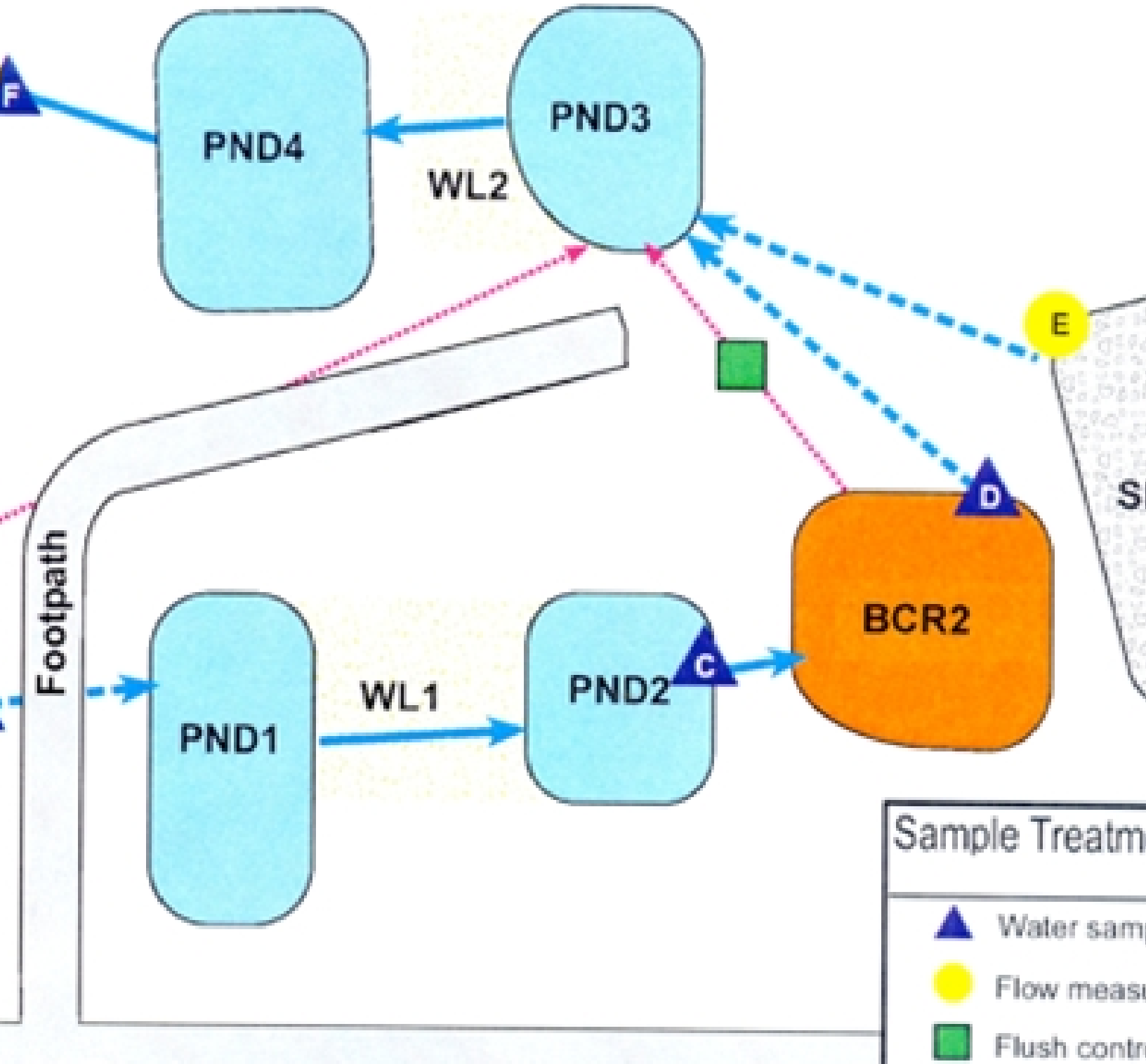

F
PND4
PND3
WL2
E
D
BCR2
Footpath
PND1
WL1
PND2
C
Sample Treatm
Water sam
Flow measu
Flush contr

Figure 6-3. Site treatment layout, example schematic.

Source: Modified from Western Pennsylvania Coalition for Abandoned Mine Reclamation, http://www.wp-camr.org/facts/groups/omr-plan/omr-schematic.pdf.

The site location should be depicted on a map and latitudinal and longitudinal coordinates should be provided, if available. Directions to the site should be provided, and the name, address and phone number of the operator should be available on site for issues regarding access and in case of emergency. Any hazards at the site should be clearly identified.

6.5.1 Site Background

The O&M Plan should identify the mine discharge and summarize its history, if known. The effluent chemistry and flow rate should be described. Changes in flow rate or chemistry over time should be described.

The O&M Plan should identify the receiving stream of the discharge. Any known sources of mining-influenced or other contaminated water flowing into the receiving stream should be identified within one mile upstream or downstream of the discharge point. The receiving stream quality prior to installation of the MIW treatment system should be described. When describing the data, the location of in-stream sampling points should be identified.

6.5.2 Treatment System

The O&M Plan should describe the treatment system's design and construction, as well as identify costs for operation and maintenance activities.

Each major system component should be described, and the component should be assigned a unique name or identifier. A brief description of the component's function in the overall treatment system should be provided. Any unique features or aspects of component installation should also be described.

An overall description of the flow system should be provided, and each of the flows entering or leaving the system should be described. Specific flow data from pilot testing or start-up should be included when available. The maximum design flow rate should be specified, and emergency spillways as well as other flow control or erosion control features should be identified.

A description of the system's overall treatment process should be provided. The system performance parameters, including expected changes in pollutant concentrations, flow rates and other performance parameters should be described.

6.5.3 Flow Measurement and Water Sampling

Control of flow within the treatment system is essential to maintaining proper system function. Therefore, flow rates should be monitored wherever water enters or exits the system. Flow should be monitored before and after system components that may become clogged or lose permeability, in order to identify problems.

The O&M Plan should identify all flow measurement points in the system, the frequency of sampling and the method of flow measurement (for instance weir or bucket and stopwatch). If a weir is used, include a table or chart that convert water levels into flow measurements. For water level measurements, a built-in scale inscribed on a wall is preferred. Otherwise measurement should be made from a clearly located datum/location that is defined in the O&M plan.

The O&M Plan should identify all locations where field tests are to be performed and where water samples are collected for laboratory analysis. Sample collection methods for each type of sample should be described, including the parameters to be measured, the frequency of measurement, the appropriate containers, and preservation and storage requirements. For each sampling point, include the parameters to be measured, the frequency of measurement, appropriate containers, and preservation and storage requirements. Wherever possible, describe field tests that may serve as substitutes for laboratory tests.

6.5.4 Maintenance Activities

Passive treatment systems require regularly scheduled maintenance. For every task the system requires the O&M Plan should identify how often the task should be performed and provide instructions. For complex tasks such as system flushing, the O&M Plan should include diagrams that identify valves and control structures. The O&M Plan should specify the content of reports that document regular inspections.

Regular maintenance includes system flushing, clearing debris, and other activities necessary to control litter, vegetation, and erosion. Valves and other movable components should be exercised. If pumps or other motorized systems are a component of the system, the O&M plan should specify necessary inspection and maintenance procedures. Such activities should be included during a complete inspection of the system at least annually, though some likely need to be addressed more frequently. Some conditions such as heavy rains, the presence of animals, drought, or small acts of vandalism require special maintenance. Identify such conditions and the non-routine activities required to keep the system operating well.

The O&M Plan should include a section describing the eventual need for large-scale maintenance and replacement of system components. Damage to the system including vandalism and metal clogging, and maintenance activities including sludge excavation, substrate replacement and limestone addition are examples of activities that can be time-consuming and costly. When possible, predict the time frame for conducting major maintenance activities, and identify associated costs and funding mechanisms.

6.5.5 Health and Safety

Every BCR system should have a site specific health and safety plan (HASP) that advises inspectors of the hazards of H_2S. Liability issues prevent ITRC from including a HASP in this guidance document. The HASP should be prepared by a licensed professional familiar with the hazards.

7.0 TECHNICAL AND REGULATORY CHALLENGES AND SOLUTIONS

Although BCRs can remove metals or other constituents of concern from MIW, in many instances their performance has not been perceived as consistently repeatable from site to site. This perceived lack of consistency can be overcome through careful planning, engineering, and design. Proper attention must therefore be paid to the site-specific technical aspects of a project and regulatory requirements to determine the potential applicability of BCRs.

7.1 Technical Challenges

MIW is generally a result of uncontrolled releases of constituents of concern from historical operations over large or remote areas. Because of changes in regulations, releases from new mines are not as common or extensive as releases from historical mining operations. Table 7-1 presents some of the typical technical issues that can be encountered and references to specific sections within this document that address these issues.

Table 7-1. Technical challenges and guidance for these challenges

Technical Challenges	Referenced Sections
Site Locale and Space	
Discharge sites for MIW, particularly for abandoned mines, can often be in isolated areas of steep, rough terrain that are difficult to access and may not have electricity. Space for the placement of a BCR can often be a challenge in narrow valleys, steep hillsides, or when additional private land may be required but not available for use.	• Section 2.6: Evaluate Site Environment and BCR Size • Section 4.0: Design • Section 5.2.1: Physical Settings
Site Locale—Elevation and Climate	
Higher elevation and colder climates can reduce year round performance due to lowered microbial activity, however insulating BCRs can help compensate. In areas with intense rainfall events or large spring flows, equalization basins may be required for storage of larger volume flows and diversion of clean water around treatment system.	• Section 2.1: Characterize Influent • Section 2.6: Evaluate Site Environment and BCR Size • Section 4.2.3: Temperature Effects • Section 4.2.4: Flow variability • Section 5.2.2.5: Cover Installation • Section 6.1: System Start-up • Section 6.2.2.2: Water Chemistry and Quality Parameters
Site Locale—Neighboring Land Use	
A vital aspect of site locale that should be considered is the proximity and compatibility to neighboring land uses, principally residential areas, where local acceptance is important. The aesthetic properties, in addition to odor and safety issues for humans and wildlife, should be recognized and managed.	• Section 2.9: Public and Tribal Concerns and Acceptance • Section 4.0: Design • Section 4.5: Site Control Design • Section 6.2.1: Monitoring the Physical Condition of the System
Hydrogen Sulfide Gas—Odor and Toxicity	
Hydrogen sulfide gas is produced in BCRs in varying levels. The designer must ensure that surrounding res-	• Section 2.0: Determining the Applicability of a Biochemical Reactor

Technical Challenges	Referenced Sections
idences or work places are not exposed to toxic levels of hydrogen sulfide gas. Low levels of hydrogen sulfide gas emissions may be below the toxicity level but can still result in nuisance odors (rotten egg smell) to surrounding properties.	• Section 2.3: Evaluate Treatment Effectiveness for Contaminants of Concern • Section 2.9: Public and Tribal Concerns and Acceptance • Section 4.0: Design • Section 5.2.2.5: Cover Installation • Section 6.2.1: Monitoring the Physical Condition of the System
Aesthetic Issues	
Often the appearance of the BCR and pre- and posttreatment cells concerns surrounding land users. Some BCRs are designed to look like wetlands areas and some are designed with plastic covers. Consider public acceptance of the site's appearance when building BCRs in populated areas.	• Section 2.9: Public and Tribal Concerns and Acceptance • Section 4.5: Site Control Design
Accessibility, Trespassing, and Safety	
The accessibility of the site can influence the likelihood of trespassing and public safety. The designer must carefully consider the use of open ponds in residential areas that may attract children or curious adults. Open ponds that are lined with plastic liner can attract animals as well and may create trapping and drowning hazards for animals and humans. Designs can be modified to limit access and prevent trapping hazards for humans and animals.	• Section 2.0: Determining the Applicability of a Biochemical Reactor • Section 2.9: Public and Tribal Concerns and Acceptance • Section 4.5: Site Control Design • Section 5.2.1.1: Siting the BCR System • Section 5.2.4: BCR Systems Site Construction Restoration • Section 6.2.1: Monitoring the Physical Condition of the System
Mine Influenced Water Quality	
The size of a BCR is a function of metal loading rate, which includes factors such as flow, metals concentration and required retention time. A higher metal loading rate requires a larger system and more retention time, which can be a challenge when adequate space is not available. The MIW quality can necessitate a pretreatment or posttreatment step(s).	• Section 2.1: Characterize Influent • Section 2.2: Define Treatment Goals • Section 2.3: Evaluate Treatment Effectiveness for Contaminants of Concern • Section 2.5: Determine Pre- and Posttreatment Requirements • Section 4.2.1: Influent Chemistry Concerns • Section 4.4.2.4: Overflow Structures and Loading Control
Substrate Issues	
Local availability and cost of substrate can be a challenge to the siting and construction, as well as the effectiveness, of BCR treatment. Substrate type can also affect the amount of operation and maintenance including removal, replacement, and disposal–either on site or off site.	• Section 2.7: Evaluate Substrate • Section 2.8: Evaluate Cost • Section 3.1: Selecting BCR Substrate • Section 5.2.2.4: Substrate Installation • Section 7.2: Legislative and Regulatory Challenges
Maintenance	
BCRs requiring regular chemical input or a power source will require more frequent operations and maintenance activities than passive BCRs. A key aspect of a successful BCR is to plan adequately for expected main-	• Section 6.3: Maintenance • Section 6.5: Contents of an Operation and Maintenance Plan

Technical Challenges	Referenced Sections
tenance issues and identify who will be responsible for maintaining the system in perpetuity.	
Troubleshooting	
The long-term success of a BCR depends on identifying and resolving issues preventing effective treatment of MIW. The two primary categories of troubleshooting problems with a BCR include chemical trends and physical trends.	• Section 4.4.4: Bed Clogging • Section 4.4.5: Short Circuiting • Section 6.2.1: Monitoring the Physical Condition of the System • Section 6.3.2: Troubleshooting

7.2 Legislative and Regulatory Challenges

Often, there are many permitting issues that must be addressed when constructing treatment systems, from land disturbance and erosion and stormwater control issues to discharge permits for active mining operations. BCRs have applicability across the country at abandoned and active mining sites; however, local, tribal, and state and federal regulations may differ for each site. Federal requirements are discussed on the National Pollutant Discharge Elimination System (NPDES) web site mining page (USEPA 2012). All required permitting agencies should be consulted throughout the process to ensure that appropriate regulatory processes are being followed. Note that the following discussion does not include all of the existing regulatory programs in every state.

Most of the early BCRs were used on abandoned or inactive sites. In these cases, the receiving streams were often devoid of aquatic life. The goal of treatment was to improve water quality in the stream so that it could once again support an aquatic community. As a result, the key issue was usually increasing the pH and reducing the metal toxicity of the discharge. No strict numeric standards were imposed. A well-designed passive treatment system will increase pH and remove metals, but may not be able to continually meet strict numeric limits. Pennsylvania still allows passive systems, including BCRs, to be used at abandoned sites without permit restrictions. These are considered to be more like a best management practice, rather than a permitted facility. The West Fork BCR system (see Appendix B.7) in Missouri is operated under an NPDES permit. Regulatory challenges at this site are discussed in Gusek et al. (1998).

As the technology has matured, there are a growing number of applications where the goal is to meet NPDES discharge standards. Although BCRs can generally meet pH limits, meeting very low chronic toxicity standards can be problematic. However, the biotic ligand model, which accounts for reductions in toxicity when metals are complexed with organic carbon typically found in BCR effluent, may be worth examination in the context of BCR permitting. In addition, the initial water from the BCR can contain high levels of organic carbon and nutrients. These parameters may require some form of management or treatment prior to discharges as described in Section 2.5 and Section 4.3.5.

7.3 Disposition of Residual Materials

As with all residual materials, the designation of waste must consider if the material still has economic value. In many cases residual materials from BCRs can include high concentrations of

metals that may prove to be economically recoverable (Gusek and Clarke-Whistler 2005; Gusek, Wildeman, and Conroy 2006). Use care in appropriately evaluating and managing any residual material removed from a BCR. In the federal code, 40 CFR 261.24 lists the maximum concentration of contaminants for the characteristic of toxicity from the toxicity characteristic leaching procedure (TCLP) test that would define a substrate as hazardous waste or not. Nevertheless an in-depth review of all materials removed from the system routinely or at the end of its service life should be done to establish proper disposition (see Section 2.5.3 and Section 6.3.1).

7.4 Wetlands (CTW and Mitigation)

One of the most pressing challenges in some areas is how to address wetlands present at a site. Differences exist between United States Army Corp of Engineers (USACE) districts as to the applicability of §404 of the Clean Water Act (CWA) to wetlands that were created by mine drainage. In order to create treatment systems for MIW, occasionally, poor quality wetlands created by the MIW must be removed. Consult with the local USACE district and other agencies at the planning stage to determine regulatory requirements regarding any wetlands that may be present and any mitigation (in-kind, out-of-kind, on-site, or off-site) that may need to be performed. Additionally, tribal, state, and local agencies should also be consulted about wetland permitting and mitigation issues. For additional guidance, see Chapters 7 and 8 of ITRC's WLTD-2 document (ITRC 2005).

Mayer Ranch Wetlands Mitigation

At the Mayer Ranch Superfund site, mitigation of the volunteer cattail wetlands that sprung up after the mine water began to discharge in 1979 was not required. In theory, the clean polishing pond at the end of the treatment system could replace (in terms of mitigation credits) the contaminated volunteer wetlands that were destroyed during construction of the treatment system.

8.0 STAKEHOLDER CONCERNS AND ISSUES

Community stakeholder concerns for BCRs used to treat mining-influenced water (MIW) revolve around three major points:

- potential health effects and nuisance odors due to proximity
- inability to improve the quality of MIW due to strict adherence to environmental regulations
- inability to beneficially use MIW due to water quality concerns and water rights laws in individual states

Citizens and volunteer groups have a profound interest in improving the damage from past mining operations and preventing future damage during existing mining. Citizen groups have been formed specifically to help monitor effluent from MIW (including BCRs) treatment facilities but have encountered regulatory obstacles (for example, third party liability and National Pollution Discharge Elimination System (NPDES) permitting) that offer more financial risk than these groups can accept. Federal Good Samaritan legislation could reduce many obstacles to improving water quality that communities and industry face.

8.1 BCR Site Location

For landowners and public stakeholders, the proximity of BCRs and any other treatment system for MIW to populated areas must be given consideration. The public is generally supportive of methodologies that clean up mine drainage and restore the environment. The difficulty arises, at times, with the placement of treatment systems. When planning BCRs, consider the impacts that BCR placement or operations will have on neighbors. Generation of offensive odors as a result of treatment, noise from equipment during and after construction, and the overall appearance of the treatment system should be topics discussed with landowners near the site.

8.2 Abandoned Mine Drainage

In many areas of the country, MIW is the result of unregulated mining activities of the past, which pollute streams and, in some cases, drinking water supplies with metals contamination, acidity, or both. With the advent of watershed groups in the late 20th century came the desire of these local non-profits to clean up degraded streams in their areas. In areas of the country with a history of mining, the remediation of MIW became a primary focus. These volunteer groups, located in the western hard rock areas and in the eastern coalfields, began to secure funding and technical assistance to help address MIW. These local community groups are finding it increasingly difficult, however, to construct treatment systems due to concerns about funding, permitting, liability, and legislative issues.

8.2.1 Funding for Abandoned (MIW) Mine Drainage Treatment

Adequate funding for construction of MIW treatment systems (including BCRs) has generally been lacking throughout the country. Despite the presence of watershed groups able and willing to operate and maintain treatment systems in metal mining areas of the west, few funds exist to assist them. Presently, Clean Water Act (CWA) Section 319 program funds often cannot be used at these western sites because many of the discharges are considered point-source discharges, thereby rendering them ineligible for 319 funding.

In the eastern coalfields, watershed groups have long addressed abandoned coal mine discharges using Section 319 funds, since MIW in these areas is considered non-point source pollution by USEPA. However, in spite of being able to access and use Section 319 funds to address MIW in the east, the need for funding far outweighs available Section 319 funds.

No mechanism is currently in place to generate funding from active hard rock mining to address abandoned hard rock mine drainage remediation activities (as there is with abandoned coal mine drainage). Mining activities in the East (coal mining) versus the West (hard rock, metal mining) are guided by two separate pieces of legislation. Hard rock mines are covered under the 1872 Mining Law while coal mining activities, both active mining and abandoned mine reclamation, are covered under the Surface Mining Control and Reclamation Act (SMCRA).

Even though abandoned mines, and the desire to clean them up, are shared amongst eastern and western communities, remediation and reclamation activities for each region's abandoned mine problems proceed very differently.

8.2.2 Permits for Abandoned Mine Drainage Treatment

Permitting of passive treatment systems sometimes may be described with Voltaire's statement "the perfect is the enemy of the good." Regulations often stipulate a perfect solution, but a citizen volunteer group may consequently abandon a good solution as a result of these requirements.

The issuance of a NPDES permit is required under federal regulation 40 CFR part 122, but the type of effluent limits that this permit includes are at the discretion of tribal or state agencies or USEPA. In the case of abandoned mine discharges, both in the western metal mining states and in the eastern coalfield states, federal, tribal, and state agencies have not given a single clear message as to whether these projects need permits and, if so, what standards those permits will require. The regulatory point of view often is to protect the water body to the extent possible and to defer permit issuance until construction is completed.

In cases where a volunteer organization attempts to implement an MIW treatment system, an undefined future permit may preclude the entire project. Violation of permit limits can have significant civil or even criminal impacts, while doing nothing has no such consequences. Although passive treatment systems are very effective at significantly reducing pollution and restoring streams, meeting a strict permit is more uncertain. It is often difficult for passive treatment systems

to consistently meet stringent pollutant discharge limits. Additionally, the increased cost and personnel demands that could be associated with monitoring and reporting requirements included in NPDES permits may be challenging for many volunteer organizations to meet.

8.2.3 Challenges for Nonprofits and Industry

In a number of instances, "Good Samaritans" in the hard rock areas of the west have been willing to conduct mine reclamation at these sites if they have environmental liability protection from the CWA and perhaps the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA). Watershed groups working on abandoned mine sites in the eastern coalfields have been conducting remediation projects with little to no protection from liability under the CWA.

The principle of primary jurisdiction, which applies when a court and an administrative agency (such as a state environmental agency) have concurrent jurisdiction, is the principle that a court will first allow that administrative agency to decide an issue, prior to the court ruling on a matter. The result is that the court, on behalf of a third party, will not compel additional or different action by a responsible entity while that entity is actively working with a state environmental agency to implement a work plan. In practice, this principle can protect a nonprofit that may be voluntarily working with a state agency to address MIW from liability for the discharge; however the protection lasts only until the work plan is complete. At that point, the owner or operator becomes liable under the CWA for the MIW discharge, and could be subject to enforcement action by third parties or the state, federal, or local environmental agency. Proponents of Good Samaritan legislation have stated that an amendment to the CWAis one possible vehicle for Good Samaritan protection. Because of the broad wording of the CWA, complete protection from liability can only occur through additional federal legislation.

For the eastern coalfield watershed groups, the interest in, and need for 'Federal Good Samaritan' protection stems from recent court decisions in Federal court in West Virginia. Although those court decisions addressed bond forfeiture sites, not abandoned sites as defined in the Surface Mining Control and Reclamation Act, the wording in those court decisions puts watershed groups operating treatment systems for Abandoned Mine Drainage in danger of being treated as industrial dischargers, having to maintain discharge standards even though they were not the original party responsible for the pollution. The opinion (West Virginia Highlands Conservancy v. Huffman, 4th U.S. Circuit Court of Appeals, No. 09-1474, 8/11/10) stated in part:

> "The text of the CWA, as well as the corresponding regulations issued by the Environmental Protection Agency, confirm that the permit requirements apply to anyone who discharges pollutants into the waters of the United States. Under the CWA, it does not matter that a mining company may have created the conditions that call for reclamation. What matters is that an entity, private or public, is currently discharging pollutants into the waters of the United States."

While the need for Good Samaritan legislation is well recognized, there are a number of opinions as to how broad the liability protections should be and to whom they should apply, in addition to how an operator would qualify for such protection. A lack of consensus on these issues has

prevented federal legislation from moving forward. National groups such as Trout Unlimited and regional groups such as the Animas River Stakeholders Group in the west and The Western Pennsylvania Coalition for Abandoned Mine Reclamation in the eastern coalfields continue to work with the USEPA and other interested parties toward a solution.

The following paper, *Good Samaritan Legislation Will Help Solve the MIW Problem*, has been presented to the ITRC Stakeholder group for review and follow-up.

Federal Good Samaritan Legislation Will Help Solve the MIW Problem

Long before the advent of modern environmental regulations, mining operations closed down when mining ceased and the operators essentially walked away from mine sites. These sites have been discharging Mine Influenced Water (MIW) for, in some cases, well over one hundred years. In the western hard rock mining states and in the eastern coalfields, discharges from these abandoned mine operations remain a huge environmental problem. Acidic metals-laden MIW, leaches into streams, killing fish and other organisms, and threatening human health and drinking water supplies. In some cases, the health risks for tribes may be greater when the tribe doesn't own the land where the discharge is located but has fishing rights in the receiving stream.

The problem is larger than the tribal, State, and Federal agencies assigned to tackle the problem. In spite of this, there are local environmental and watershed groups who wish to clean up these discharges, restore the land, and create a safer environment for future generations. The concerned citizens who volunteer for these local groups had no role in creating the pollution that causes these toxic discharges. Logically, they should not be held responsible or liable for these pollution sources. Unfortunately, the CWA never accounted for the existence and good work of nonprofit groups, community groups, tribal, State, agencies, and other Good Samaritans in cleaning up MIW from abandoned sites and has no exemption for reclamation efforts.

At more recently mined sites where a responsible party has been identified, the federal government has tools that help them hold the responsible party accountable for cleanup. At the long-abandoned sites, where a responsible party no longer exists or has not been identified, cleanups are less likely to happen. Good Samaritans play an important role at these abandoned sites. Unfortunately, current environmental laws designed to regulate active mining are of limited effectiveness when applied to abandoned mine sites.

Trout Unlimited's (TU) attempt to clean up the Tiger Mine site in Lake County, Colorado is a good example of a Good Samaritan cleaning up an abandoned mine discharge. TU constructed a BCR to clean up the discharge from the mine. Unfortunately, the BCR constructed by TU is regulated as a point source of pollution and they must obtain a clean water permit. This system, while significantly improving water quality in the mine effluent, is not likely to meet the stringent clean water requirements stipulated by the permit. Additionally, TU cannot afford the liability that comes along with a permit. In other words, this bioreactor, operated by Good Samaritans, is seen in the same regulatory light as a

present-day industrial pollution source. As a result of permit requirements and the liability issues, the bioreactor is not yet online.

It is important for regulatory agencies to take advantage of existing regulatory flexibility in providing Good Samaritan protection for organizations or agencies cleaning up MIW from abandoned mine sites. USEPA continues to investigate their administrative tools to provide meaningful assistance and protection to Good Samaritans. However, it remains unclear if it will be able to provide the full protection needed.

Although, organizations such as the Western Governors' Association (WGA) have crafted resolutions in support of a federal environmental Good Samaritan Clean Watershed Act (http://water.epa.gov/action/adopt/legislation.cfm#) for years, the issue remains unresolved and watershed groups, tribes, state agencies, and other organizations who operate or plan to construct and operate treatment systems face risk and uncertainty.

It is important for regulatory agencies to take advantage of existing regulatory flexibility in providing Good Samaritan protection for organizations or agencies cleaning up MIW from abandoned mine sites. USEPA continues to investigate their administrative tools to provide meaningful assistance and protection to Good Samaritans. However, it remains unclear if it will be able to provide the full protection needed.

8.3 MIW as an Emerging Resource

In the eastern coalfields, the recent acceleration of unconventional gas drilling activities in the Marcellus and Utica shale, and the need for water for gas extraction activities, have led the gas industry, watershed groups, states and tribes to begin examining the potential use of MIW for hydrofracking. The mine pools in the eastern coalfields are being evaluated as a potential water resource for the gas industry. However, the issue of elevated sulfates in MIW continues to be of concern to the unconventional gas industry. An additional consideration in using MIW for hydrofracking is the question of ownership. Industry is hesitant to build treatment systems for MIW discharges without the benefit of working with non-profit groups for fear of 'owning' the discharge and the accompanying liability, in perpetuity. Often, watershed groups in the east are already operating existing treatment systems. In such cases, the gas industry could then purchase the discharge water from the watershed group, who then places those funds into an operation and maintenance account for perpetual treatment of the discharge long after the gas industry has no further need for the water.

BCRs, as they relate to water quality in the eastern coalfields/shale gas area, may have a two-fold impact. First, the technology may help reduce sulfate concentrations in MIW to levels that may foster use by the gas industry, thereby ensuring the continued remediation of MIW throughout watersheds in the eastern coalfields. Secondly, eastern coalfield watershed groups have begun to use BCR-treated MIW to operate micro-hydro generators to produce electricity. As this potential use and the associated technology continue to evolve, many issues need to be addressed. These

issues include discharge water quality as it relates to fouling generation equipment and piping and coordination with federal, tribal and state permitting agencies with regard to using MIW to generate electricity.

The historically complex issue of water rights should be considered if water from a BCR or any other treatment system is being considered for use. This is an issue for both east and west as water rights are managed to a great extent under tribal or state law. In the eastern U.S., water rights generally follow the English common law principle of riparian water rights, that those owning land adjacent to water have a right to make reasonable use of that water, with amounts related to the frontage on the water body. In the west, water rights more often derive from the Colorado Doctrine of first use (Schorr 2012). In this appropriation-based system, the right to the water, and thus ownership of an annually determined amount of flow, belongs as property to whoever first made beneficial use of the water body.

9.0 TRIBAL CONCERNS

Most Native American tribes differ from western European or modern American societies in their perception of environmental protection and remediation. Therefore, it is important to consult the tribes that may be affected by a proposed BCR before beginning even the conceptual design. Federally recognized tribes and USEPA have jointly developed consultation policies and procedures for use before initiating environmental regulations or actions that may affect tribes. If you are proposing BCRs on or near tribal lands or in areas of tribal jurisdiction, contact the appropriate USEPA regional office and ask for the USEPA tribal liaison to help guide you on the appropriate tribal consultation procedures.

9.1 Regulatory Challenges

Most tribes have established environmental departments, which are funded primarily by USEPA. As with states, USEPA can delegate authority to tribes for environmental programs, including regulatory authority under the Clean Water Act, if the tribe applying for such authority has an established environmental department and satisfies other USEPA requirements. Consequently, many tribes have developed enforceable water quality standards. Although the Quapaw tribe has developed its own water quality standards, these standards are not enforceable until the tribe receives programmatic authority from USEPA. The tribe has been in close communication, however, with those designing the Beaver Creek area BCR regarding the tribe's water quality standards which reflect the tribe's priorities in addressing surface water contaminants. Although the BCR may have the effect of treating water that meets Oklahoma Water Quality Standards, the Quapaw tribe may not view the resulting water quality as being protective of the tribe, considering the tribe's more intensive use of water and water-related resources (see Appendix B.1, Beaver Creek case study).

9.2 Tribal Challenges

As an example of a specific tribal challenge, BCR construction is currently proposed on Quapaw tribal land that is culturally and historically significant to the tribe. The tribe refers to this land as the "Catholic 40" property. In the late 1800s, the Catholic Church built an Indian school and church on this property so that Quapaw children could attend the local school instead of being sent to far away Indian schools, as was the Bureau of Indian Affairs policy for children of other tribes. A tribal cemetery is also located on this property, and tribal members practice traditional subsistence and cultural gathering of plants along Beaver Creek in this area. Finally, Beaver Creek flows through the Quapaw tribe's pow-wow grounds, immediately downstream of the proposed BCR. Tribal members gather various food and ceremonial resources along Beaver Creek during the annual pow-wow, and the creek plays an important part in their cultural and religious practices.

Given the central role of Beaver Creek in tribal life, it has been crucial to seek input from the tribal leaders and the tribe's environmental department early in the conceptual design of the BCR and

well before any field investigations. For the proposed Beaver Creek BCR, the following step-by-step procedure was followed to incorporate tribal input into information and data gathering:

1. USEPA encouraged Dr. Robert Nairn of Oklahoma University to contact the Quapaw Tribe Environmental Department to discuss the possibility of beginning data gathering for investigating the feasibility of a BCR on the Catholic 40 property in the Beaver Creek area.
2. Tim Kent, Quapaw Tribe Environmental Director, approached Quapaw tribal leaders to explain the general concept of a BCR (using the Mayer Ranch BCR as an example; see Appendix B.2, Mayer Ranch case study) and to inquire about the possibility of collecting the data necessary to adequately evaluate the feasibility of constructing and operating a BCR in the Beaver Creek area.
3. After asking many questions regarding the effects of the BCR and its construction and maintenance, the Quapaw Tribal Business Committee reluctantly approved the initiation of data gathering and monitoring related to the BCR feasibility study, contingent on the assurance that no historic or culturally significant areas or features would be disturbed during the data collection activities. The Quapaw, like many Native American tribes, believe that humans cannot "fix" Mother Earth once they have severed their sacred connection with her; therefore, the tribe is very skeptical about the value of a BCR.
4. The Tribal Historic Preservation Officer (THPO) assessed and mapped historic and culturally significant features on the Catholic 40 property, where the sampling and construction related to the proposed BCR is planned.
5. The THPO and the tribe's Environmental Director assured the tribe that sampling and construction related to the proposed BCR would not adversely affect any historic or culturally significant features. With these assurances, the Tribal Business Committee granted approval to begin designing the BCR. Once complete, the draft design will be presented to the Tribal Business Committee for approval.

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APPENDIX A. PHYSICAL, CHEMICAL, AND BIOLOGICAL REACTIONS IN A BIOCHEMICAL REACTOR

A.1 Chemical Reactions and Equilibrium in BCRs

The set of reactions of interest in a BCR are dominated by the interaction of the substrate, organisms in the substrate, and the water flowing through that substrate. Because BCRs may be applied to waters ranging widely in pH and containing any of a suite of metals, the following information cannot be exhaustive; instead, the most commonly encountered mechanisms are presented in terms of various contaminants of concern. It is advised to use chemical equilibrium software such as MINTEQ, PHREEQC, or Geochemist's Workbench to determine the likely species for a specific water composition as it is subjected to changes in pH and redox potential (Eh). Equilibrium is the driving force for reactions, but equally important are the kinetics which determine the rate of the reaction. Although a reaction may be predicted from equilibrium, the rate might be negligible for reasons such as activation energy or the requirement for diffusion of reactants through boundary layers or solids.

The set of reactions identified as potentially important in BCRs treating MIW is shown in Table A-1.

Table A-1. Geo- and biochemical reactions potentially occurring in BCRs treating MIW

Reaction	Stoichiometry	System redox & pH
Iron Oxidation	$4\ FeS_2(s) + 14\ O_2 + 4\ H_2O \rightarrow 4\ Fe^{2+} + 8\ SO_4^{2-} + 8H^+$ $4\ Fe^{2+} + O_2 + 4\ H^+ \rightarrow 4\ Fe^{3+} + 2\ H_2O$ $4\ Fe^{2+} + O_2 \rightarrow 4\ Fe^{3+} + 2\ O_2^-$ $2\ O_2^- + 2\ H_2O \rightarrow 4\ OH^-$ $2\ FeS_2(s) + 14\ Fe^{3+} + 8\ H_2O \rightarrow 15\ Fe^{2+} + 2\ SO_4^{2-} + 16\ H^+$	Aerobic, decreases pH
Carbonate System	$CO_3^{2-} + H^+ D\ HCO_3^-$ $HCO_3^- + H^+ D\ H_2CO_3$ $H_2CO_3 D\ H_2O + CO_2$	Both, increases pH
Calcium Carbonate Dissolution	$CaCO_3(s) \rightarrow Ca^{2+} + CO_3^{2-}$	Both
Water Dissociation	$H_2O\ D\ H^+ + OH^-$	Both
Iron oxide precipitation	$Fe^{2+} + 2\ H_2O \rightarrow Fe(OH)_2(s) + 2\ H^+$	Anaerobic, High pH
Iron oxide precipitation	$Fe^{3+} + 3\ H_2O \rightarrow Fe(OH)_3(s) + 3\ H^+$ $Fe(OH)_2(s) \rightarrow FeO(s) + H_2O$ $Fe(OH)_3(s) \rightarrow FeO(OH)(s) + H_2O$ $2\ FeO(OH)(s) \rightarrow Fe_2O_3(s) + H_2O$	Aerobic
Iron carbonate	$Fe^{+2} + CO_3^{2-} \rightarrow FeCO_3(s)$ $Fe^{+2} + CaCO_3(s) \rightarrow FeCO_3(s) + Ca^{+2}$	Both
Iron sulfide	$Fe^{+2} + S^{2-} \rightarrow FeS(s)$	Anaerobic
Aluminum oxidation	$Al^{3+} + OH^- D\ Al(OH)^{2+}$ $Al(OH)^{2+} + OH^- D\ Al(OH)_2^+$	Both, precipitates at higher pH

Reaction	Stoichiometry	System redox & pH
	$Al(OH)_2^+ + OH^- \rightarrow Al(OH)_3(s)$	
Divalent metal carbonates	$M^{2+} + CO_3^{2-} \rightarrow MCO_3(s)$	Both, higher pH
Divalent metal hydroxides	$M^{2+} + 2\,OH^- \rightarrow M(OH)_2(s)$	Both, higher pH
Divalent metal sulfides	$M^{2+} + S^{2-} \rightarrow MS\,(s)$	Anaerobic
Divalent metal adsorption by organic	$M^{2+} + 2$ R-COO- → R-COO-M-OOC-R	Both
Divalent metal adsorption by iron oxide	$M^{2+} + 2$ -Fe-O- → -Fe-O-M-O-Fe-	Aerobic
Cellulose degradation yielding glucose	$(C_6H_{10}O_5)_n + H_2O \rightarrow (C_6H_{10}O_5)_n + C_{12}H_{22}O_{11}$ $C_{12}H_{22}O_{11} + H_2O \rightarrow 2\,C_6H_{12}O_6$	Both
Glucose fermentation by cellulose degraders	$C_6H_{12}O_6 \rightarrow 2\,CH_3CH_2OH + 2\,CO_2$ $C_6H_{12}O_6 + 4H^+ \rightarrow 2\,CH_3CH_2COOH + 2\,H_2O$ $C_6H_{12}O_6 + 2\,CO_2 + 4\,H^+ \rightarrow 2$ HOOC$(CH_2)_2$COOH $+ 2\,H_2O$	Anaerobic
Organic electron donation (glucose shown)	$C_6H_{12}O_6 + 6\,H_2O \rightarrow 6\,CO_2 + 24\,H^+ + 24\,e^-$	Both
Hydrogenotrophic electron donation	$H_2 \rightarrow 2H^+ + 2e^-$	Both
Respiration	$4\,H^+ + 4\,e^- + O_2 \rightarrow 2\,H_2O$	Aerobic
Denitrification	$12\,H^+ + 12\,e^- + 2\,NO_3^- \rightarrow 5\,H_2O + OH^- + N_2$	Anoxic
Iron reduction	$H^+ + e^- + Fe^{3+} \rightarrow Fe^{2+} + H^+$	Anaerobic
Sulfate reduction to sulfide	$8\,H^+ + 8\,e^- + SO_4^{2-} \rightarrow 2\,H_2O + 2\,OH^- + H_2S$	Anaerobic
Methanogenic electron acceptance	$8\,H^+ + 8\,e^- + CO_2 \rightarrow CH_4 + 2\,H_2O$	Anaerobic
Table inspired by Klusman et al. (1993) Although all reactions shown are reversible, only the dominant direction in BCRs and MIW treatment is shown in most cases.		

A.1.1 pH and Alkalinity

Mine sites with neutral or alkaline mine water are uncommon and generally reflect a lack of iron. Pyrite (FeS) is most frequently the source of acid mine drainage. The decrease in pH is due to the series of reactions, most often biologically-driven, giving net sulfide oxidation. The specific set of reactions begins with sulfide oxidation:

$$4\,FeS_2(s) + 14\,O_2 + 4\,H_2O \rightarrow 4\,Fe^{2+} + 8\,SO_4^{2-} + 8H^+$$

The ferric iron (Fe^{2+}) is slowly oxidized at low pH to ferrous iron (Fe^{3+}):

$$4\,Fe^{2+} + O_2 + 4\,H^+ \rightarrow 4\,Fe^{3+} + 2\,H_2O$$

Although this reduces H+ and should raise pH, the resulting ferric ion is able to accept electrons from pyrite with release of protons such that there is a net slight increase in acidity:

$$2\ FeS_2(s) + 14\ Fe^{3+} + 8\ H_2O \rightarrow 15\ Fe^{2+} + 2\ SO_4^{2-} + 16\ H^+$$

The cases in which mine water is not acidic most often are mines in a carbonate lithology (limestone or dolomite), as carbonates buffer water towards neutral pH. These minerals dissolve to some degree in water. For limestone ($CaCO_3$), the solubility product, K_{sp}, describes saturation:

$$CaCO_3(s) \rightarrow Ca^{2+} + CO_3^{2-}\ K_{sp} = 4.5 \times 10^{-9}\ (mol/L)^2 = [Ca^{2+}][CO_3^{2-}]$$

The K_{sp} given is in pure water at 25°C (Krauskopf 1979). If acidic water encounters dissolved carbonate, the reactions of the carbonate system result in CO_2 release. This alternately can be thought of as the reactions resulting in the science fair volcano when acid (vinegar) and carbonate (baking soda, sodium bicarbonate ($NaHCO_3$)) are mixed and carbon dioxide is rapidly released:

$$CO_3^{2-} + H^+ \rightarrow HCO_3^-$$

$$HCO_3^- + H^+ \rightarrow H_2CO_3$$

$$H_2CO_3 \rightarrow H_2O + CO_2$$

The same chemistry is used to reduce acidity in MIW, with limestone placed such that the acid contacting the limestone results in dissolution and use of the carbonate to raise pH. Note that the increase in pH and the presence of carbonate also affects metal solubility as a result of the coating of the limestone due to the formation of metal hydroxides, carbonates, and sulfates typified by green rust (variable composition and valence, layered iron hydroxides (for example, $Fe(OH)_2$ (Bernal et al. 1959), yellowboy ($Fe(OH)_3$) and siderite ($FeCO_3$).

That the release of gaseous carbon dioxide is due to neutralization in some cases can be used to raise pH without dissolving carbonate. If the mine water is supersaturated by carbon dioxide when compared to the atmospheric concentration, then carbon dioxide will be released simply because the vapor pressure of CO_2 in the solution is greater than in the air. As the CO_2 is released, equilibrium draws the above equations to the right to produce more CO_2 and thus decrease H^+ concentrations.

Another method for increasing alkalinity is biological sulfate reduction. In sufficiently anaerobic environments (Eh below about -150 mV (ref)) with an organic electron donor present, the paired electron donation and acceptance reactions are[1]:

$$CHO \rightarrow CO_2 + H_2O + H^+ + e^-$$

Electron donation from the organic substrate is coupled with acceptance by the sulfate formed from oxidation of pyrite, resulting in a number of possible products; if producing sulfide, the electron acceptance is:

$$SO_4^{2-} + 8(H^+ + e^-) \rightarrow S^{2-} + 4\ H_2O$$

[1]Reactions are not stoichiometrically balanced for any specific organic compound.

The sulfide is a component of alkalinity. At a pH below neutral, the sulfide will equilibrate to mostly reform hydrogen sulfide, thereby removing acidity:

$$S^{2-} + 2\ H^{+} \rightarrow H_2S$$

Thus sulfate reduction results overall in the generation of about two equivalents of alkalinity per mole of sulfate reduced.

A.1.2 Iron

Iron is sometimes present in MIW in large quantity, and when present is almost always in the ferrous (+2) form. Because of the large amount of iron, the decision tree for treatment generally starts with consideration of the fate of iron. If large amounts of iron will precipitate from water as it is neutralized, the BCR must be designed to handle both the chemistry and the large volume of sludge, which could clog a BCR.

The redox potential determines what reactions, if any, occur with iron. In the presence of oxygen, the following set of reactions can occur, with the extent of reaction, and product formed, depending on activity and time:

$$4\ Fe^{2+} + O_2 \rightarrow 4\ Fe^{3+} + 2\ O^{2-}$$

$$2\ O^{2-} + 2\ H_2O \rightarrow 4\ OH^{-}$$

This redox reaction is highly dependent on Eh, which is related to oxygen fugacity in water. Most waters containing iron and saturated with atmospheric oxygen will support Fe^{3+} formation when pH is above 4-5. If the oxygen content is low, iron may form iron (II) hydroxide:

$$Fe^{2+} + 2\ H_2O \rightarrow Fe(OH)_2\ (s) + 2\ H^{+}$$

In water above pH 5 and at redox values associated with dissolved oxygen being present, a series of reactions of the ferric ion involving water results in variously hydrated iron species:

$$Fe^{3+} + 3\ H_2O \rightarrow Fe(OH)_3\ (s) + 3\ H^{+}$$

Ferric ion forms iron (III) hydroxide, the commonly seen "yellowboy":

$$Fe(OH)_2\ (s) \rightarrow FeO\ (s) + H_2O$$

Ferrous oxide may be observed as an intermediate:

$$Fe(OH)_3\ (s) \rightarrow FeO(OH)\ (s) + H_2O$$

Iron (III) hydroxide converts to iron oxyhydroxide:

$$2\ FeO(OH)\ (s) \rightarrow Fe_2O_3\ (s) + H_2O$$

Iron (III) oxide also may form.

Because of the range of iron/water combinations, geochemists tend to use broad terms such as iron oxyhydroxide. The iron oxyhydroxide formed in BCRs may be amorphous or may form a number of named minerals. These minerals may include other anions such as carbonates and sulfates. Goethite (FeO(OH)) is often observed in BCRs, but polymorphs of goethite may also occur, as well as other minerals such as hematite (Fe_2O_3). In environments with limited or no oxygen, green rust comprised of various configurations of ferrous hydroxide may be found.

Iron oxidation will produce H^+ and either consume alkalinity or lower pH (for example, the net set of reactions forming goethite are $4\ Fe^{2+} + O_2 + 6\ H_2O \rightarrow 4\ FeO(OH) + 8\ H^+$). Sufficiently low pH and/or redox potential (Eh) can drive reactions backwards to resolubilize iron.

A summary of the above chemistry is that iron is easy to remove as rust if pH is above 6 and the water is aerobic. As iron oxyhydroxides form and precipitate, various other metals may be adsorbed to the surface. With continued growth, these adsorbed metals are incorporated as a co-precipitate. Therefore the formation of iron rust also will remove other metals in the mine drainage. Copper, lead, zinc, cadmium, arsenic and selenium all have been reported as substantially removed via co-precipitation. pH is a significant factor in such adsorptive co-precipitation; anionic metalloids such as selenate and arsenate are adsorbed more at lower pHs. Removal of cations such as copper, lead, and zinc is promoted by basic (pH>8) conditions (Dzombak and Morel 1990).

For cases in which BCRs are built as treatment wetlands with wetland plant roots growing into the substrate, plant roots provide oxygen and create small aerobic zones in their vicinity called oxidized rhizospheres (Conway 1940). These aerobic zones can result in the formation of iron oxyhydroxides at the root surface (Bartlett 1961), with co-precipitation also occurring. The removal of metals by plants has been explained as in some cases being largely due to this iron co-precipitation mechanism (Doyle and Otte 1997).

In addition to the formation of iron oxyhydroxides, water chemistry may result in other reactions of iron. At near neutral pH and in the presence of carbonates, siderite (iron (II) carbonate) may form:

$$Fe^{+2} + CO_3^{2-} \rightarrow FeCO_3\ (s)$$

Ion exchange with calcium carbonate, often present as limestone gravel in the BCR substrate, also can form siderite:

$$Fe^{+2} + CaCO_3\ (s) \rightarrow FeCO_3\ (s) + Ca^{+2}$$

If iron carbonate was later exposed to an elevated Eh, such as by changing from anaerobic to aerobic conditions, the siderite is driven by equilibrium considerations to form magnetite (Fe_2O_3).

In strongly anaerobic conditions where sulfide (S^{2-}) is present, iron sulfide will form:

$$Fe^{+2} + S^{2-} \rightarrow FeS\ (s)$$

Similarly to iron carbonate, exposure of FeS to oxidizing conditions results in oxidation of sulfide to sulfate, potentially chemically or microbiologically driven. This oxidation is the previously described primary causal reaction resulting in acidic mine water conditions.

A.1.3 Aluminum

Aluminum, if present, is likely to form oxides as MIW is neutralized, generally precipitating in the same zone as iron. As with iron, the resulting material is often problematic because it can form large volumes of pore-restricting sludge. Chemically, aluminum tends to be thought of in the system Al-O-H as determined by pH:

$$Al^{3+} \rightarrow Al(OH)^{2+} \rightarrow Al(OH)^{2+} \rightarrow Al(OH)_3(s)$$

This series occurs as pH rises above 4-5 (at which hydroxyl aluminum forms), with amorphous aluminum hydroxide ($Al(OH)_3$) and/or the crystalline form, gibbsite, precipitating when water is between pH 6 and 10. Most mine water, however, has significant sulfate, and aluminum species are likely to be in a form of mixed hydroxides and sulfates such as ettringite ($Ca_6Al_2(OH)_{12}(SO_4)_3{\cdot}26H_2O$). As aluminum precipitates, alkalinity (OH^-) is removed.

A.1.4 Manganese

Manganese can be a particularly difficult metal to remove in a BCR. Manganese is highly soluble as Mn^{2+} in water except at fairly high Eh and pH well above 7. Above pH 7.6 or so and in high dissolved oxygen conditions (high Eh), Mn will precipitate as an oxide (in order of decreasing Eh: MnO_2, Mn_2O_3, or Mn_3O_4) or at low Eh, as manganese hydroxide ($Mn(OH)_2$). Hui et al. (2010) showed that some portion of Mn removal in limestone beds is biological, claiming that fungi may be the dominant biological oxidizers. The pH at which solids form is also affected strongly by the presence of carbonate. At modest carbonate concentrations and in somewhat anaerobic environments, manganese carbonates form:

$$Mn^{2+} + CO_3^{2-} \rightarrow MnCO_3(s)$$

Manganese carbonates are, however, an example of rates being quite important. Manganese carbonates form metastable solutions for at least 72 hours after concentrations indicate supersaturation exists (Lebron and Suarez 1999).

Manganese does form insoluble phosphates, but the addition of phosphate in some form in BCR substrate might convert a metals problem into an eutrophication problem. In contrast to reactions with phosphate, manganese does not form sulfides that precipitate from water. Thus to remove manganese by precipitation, the mine water must be made somewhat alkaline. Buffering MIW to fairly high pH by limestone dissolution may not be possible.

A.1.5 Other divalent metals: Pb, Zn, Cd, Ni, Cu

The behavior of lead, zinc, cadmium, nickel, and copper are all similar in terms of removal mechanisms. Representing each as M^{2+}:

$$M^{2+} + 2\ OH^- \rightarrow M(OH)_2\ (s)$$

Precipitation of the metal hydroxide is possible, but only at high pH. Copper, the least soluble in terms of hydroxide, is soluble to about 5 mg/L at pH 7. Zn, the next least soluble, is saturated at pH 7 around 100 mg/L. As previously noted, though, all of these metals will co-precipitate to some degree with iron oxyhydroxide.

Carbonate formation is related to pH due to the carbonate cycle. For water in equilibrium with atmospheric carbon dioxide, the pH at which the carbonate forms for these metals is also above 7. The least soluble species is lead:

$$Pb^{2+} + CO_3^{2-}\ PbCO_3\ (s)$$

With a K_{sp} on the order of 10^{-14}, carbonate must be present at around 5 x 10^{-5} molar to result in 100 ppb Pb, which requires pH to be around 8.5 if in equilibrium with atmospheric CO_2.

There are mixed hydroxide and carbonate species which may be significant in precipitation, such as malachite, $Cu_2CO_3(OH)_2$. Assuming the pH is not basic, the three metal precipitates which are highly insoluble are sulfides, phosphates, and arsenates (sulfates of these metals are highly soluble):

$$M^{2+} + S^{2-} \rightarrow MS\ (s)$$

$$3\ M^{2+} + 2\ PO_4^{3-} \rightarrow M_3(PO_4)_2\ (s)$$

$$3\ M^{2+} + 2\ AsO_4^{3-} \rightarrow M_3(AsO_4)_2\ (s)$$

If arsenic is present, that arsenic is likely the focus of treatment rather than these other, divalent metals. Similarly, high levels of phosphate are problematic, and metal phosphates are so insoluble that water with dissolved divalent metal and dissolved phosphate must have ligands or chelators present preventing or greatly slowing the rate of metal phosphate formation. Therefore sulfides are the most likely form of chemically precipitating many metals. The K_{sp} of the metal sulfide is less than 10^{-25} for Bi, Cd, Cu, Hg, Ag, Sn, and Zn; for Co it is around 10^{-20}, and for both iron and nickel sulfide, around 10^{-19}. If 0.1 mg/L of sulfide (0.003 mM) is present, then at equilibrium the concentration of the most soluble of these metals, iron, is in solution only at a concentration calculated as a few ng/L.

Precipitation as sulfide is therefore a preferred mechanism for many metals in mine drainage. Such precipitation requires that sulfide be produced, which is the desired the role of the biological component of the BCR.

A.1.6 Arsenic

Both arsenic and selenium present challenges to treat because these metalloids behave as anions in solution. Arsenic has several oxidation states, with $^{+5}$ and $^{+3}$ being the most important in aqueous systems, forming compounds of oxygen, arsenate, and arsenite. Arsenate, AsO_4^{3-}, behaves much like phosphate (P is just above As in the periodic chart), forming insoluble compounds with many

metals. Arsenic as arsenate (As^{5+}) or arsenite (As^{3+}) is soluble at pH and Eh values seen in normal waters including most MIW. In reducing conditions, thioarsenic species may form, but if iron were present, arsenopyrite could form. The challenge of water containing arsenic is currently addressed by co-precipitation with iron and/or sorption to mineral phases.

A.1.7 Selenium

As with arsenic, selenium ($^{+6}$) and ($^{+4}$) behaves more as an anion than cation. This element is analogous to sulfate, being present in oxic water as selenate (SeO_4^{2-}) or selenite (SeO_3^{2-}). At modest Eh, Se^0(s) can form, but further reduction results in selenide, Se^{2-}, generally as HSe^-. The various anions are soluble in water. Behavior and toxicity of Se in aquatic systems are dependent on the speciation. SeO can associate with natural organic matter leading to precipitation, it may substitute for iron in FeS and form various complexes (such as $FeS_xSe_y^2$) depending on reaction conditions; however, Se sulfide complexes are less likely to re-oxidize than Se organic complexes. Removal of Se from water is a combination of processes, including adsorption (such as to iron hydroxides), and biological reduction and volatilization. Biological reduction can be promoted in BCR design by sustaining an adequate electron donor supply under saturated conditions and thereby controlling Eh in the BCR.

A.1.8 Chromium

Chromium is stable in +3 and +6 valency; Cr^{6+} also forms an oxide which behaves as an anion ($HcrO_4^-$/ CrO_4^{2-}/$Cr_zO_7^{2-}$). In reducing (anaerobic) conditions above pH 4, chromium forms Cr_zO_3 and is sparingly soluble. The +6 ion is highly toxic, so BCR design to treat chromium should focus on creating an anaerobic zone to promote the formation of such chromium (III) oxide.

A.1.9 Uranium

Uranium is stable in the +4 and +6 oxidation states. Uraninite, UO_2, is sparingly soluble between pH 5 and 11 (Casas et al. 1998). However, uranium carbonates are quite soluble. Generally uranium will precipitate when reduced towards increasing proportions of U^{6+} at low Eh (e.g. as U_4O_9) corresponding to strongly anaerobic conditions.

A.2 Ligands and Adsorption

BCRs contain a source of electron donation, organic substrate. Although thought of as ‘food for bugs’, the substrate also has a capacity to remove metals by adsorption. The organics display chemical groups such as carboxylic acids, which result in adsorption of cationic metals. If the plant-derived material is a solid, like the surface of a wood chip, then the organic is a solid adsorption media. However, some of the organic compounds are dissolved (as components of dissolved organic carbon, DOC) and/or colloidal. For example, plant-derived organic material tends to degrade to compounds such as humic and fulvic acids, and adsorption of metal to these colloids means the metal is complexed with an aqueous ligand. The DOC-bound metal remains present in the aqueous phase, but is unavailable for other removal mechanisms.

In this way, organic matter both can capture metal (adsorption to solids) and prevent metal removal (DOC-bound metals). Adsorption is not a preferred mechanism for metal removal because the adsorption sites eventually will be exhausted and the reactor thus would appear to fail prematurely. Another issue related to such adsorption is that the decay of the organic material acting as an electron donor to the BCR also results in loss of adsorption sites and therefore release of previously adsorbed metals.

Adsorption equilibrium is most often represented with Langmuir or Freundlich equations. Langmuir is based on the concept of a single-layer adsorption onto a surface with some number of sites and at high concentrations goes to a saturated value:

$$q = q_{\max} \frac{K_a C}{1 + K_a C}$$

where:

q = mass adsorbed per mass of sorbent (texts use a variety of symbols also including Q, S, W, q, and x/M), mg/g

q_{max} = mass adsorbed at complete coverage, mg/g

C = concentration of the sorbed molecule, mg/L

K_a = adsorption constant related to sorbate-sorbent affinity, L/mg.

The isotherm predicted by this Langmuir equation comes to a constant value at high concentration. In some cases this behavior is not observed, and the similarly shaped but empirical Freundlich isotherm is applied:

$$q = KC^{\frac{1}{n}}$$

where:

K_f, n = adsorption coefficients

units of K_f are $(mg/g)\times(L/mg)^{1/n}$

n is unitless

A.2.1 Degradation of Organic Substrate and Sulfate Reduction

The materials supplying electrons to sulfate reducing bacteria in BCRs are usually plant-derived, such as various hays, composts, and woody materials. Although this and the following section focus on woody materials, other solid substrates such as chitin are omitted simply because of the current paucity of available literature; further, the emerging technology of soluble organic-supplied SRBs, in which the organic supply rate is controlled, are not a focus of this document.

Woody organics are largely composed of cellulose, hemicelluloses, and lignin. Cellulose is a polymer of glucose, which is difficult to degrade because of both the specialized enzymes required to

cleave the beta (1-4) sugar bonds and because of the tight, crystalline packing between molecules caused by hydrogen bonding. Hardwoods have a large amount of cellulose, resulting in the hardness of the wood. To degrade cellulose, BCR substrates frequently include cow manure because ruminants harbor gut bacteria able to cleave cellulose to sugars; it should, however be noted that Pereya et al. (2008) showed at lab scale that inocula obtained from other BCRs gave somewhat better results than dairy manure in terms of rates observed within a few months of inoculation. Hemicellulose is another polymer of various sugars with an amorphous structure and thus is similar to cellulose in biodegradability. Softwoods have a higher content of hemicellulose and lignin than do hardwoods. Lignin is sparingly biodegradable because it is a heterogeneous, highly branched large molecule containing numerous phenyl groups.

The degradation of cellulose is largely attributed to aerobic organisms. Some protozoa and bacteria are known to degrade cellulose, generally in a mutualistic relationship with termites and ruminant animals. Very few animals produce such cellulases. Lignin is less degradable, with only a few organisms producing enzymes able to degrade the molecule. Fungi degrade cellulose not only using a cellulose enzyme but also by free radical attack, which may degrade lignin; fungi are the dominant degraders of wood (Lynd et al. 2002). This is important as most BCRs are operated to be anaerobic, and cellulosic biodegradation proceeds slowly in anaerobic environments. Lignin may not be degraded at all without oxygen. Horizontal-flow BCRs may, however, have such fungi in the upper portion of the substrate which is exposed to oxygen, and it has been speculated these organisms supply create potential electron donors for sulfate reduction at a faster rate than observed in fully anaerobic BCRs.

The presence or absence of oxygen is related to two important rates when considering BCR design and operation:

1. The rate of oxygen consumption. If the influent mine-impacted water has some level of dissolved oxygen (DO), then aerobic degradation of the plant matter by organisms sets the rate of oxygen use and therefore determines the size of the aerobic zone within the BCR substrate.
2. The rate of anaerobic plant matter fermentation. Without oxygen, anaerobic cellulose degraders transform cellulose to component sugars and then ferment that sugar to some oxidized form (CO_2, H_2O) and a volatile fatty acid (e.g. lactic acid, just as human muscle operates in anaerobic exercise). That volatile fatty acid (VFA) is then used by sulfate reducing bacteria with potential competition from fermentative organisms. The concentration of sulfate is generally high enough to not limit the rate of sulfate reduction, and therefore the rate of sulfide production is limited by the rate at which the VFA is produced.

In addition to these rates, the presence of oxygen also may cause sulfate formation. In a horizontal-flow BCR or in the advancing oxic zone of a vertical-flow BCR, sulfide oxidizing organisms such as *Thiobacillus* will use sulfide as an electron donor and form sulfate. Such metabolism impacts pH in the reverse of alkalinity provided by sulfate reduction, and may liberate previously sulfide-trapped metals.

A.2.2 Quantifying Plant Matter Degradation

Researchers examining woody degradation consider plant materials to have a biodegradable and a nonbiodegradable fraction. Measuring biological oxygen demand (BOD) of wood directly is probably impossible, so instead volatile suspended solids (VSS) or total organic carbon (TOC) is used. Either measurement can be thought of as having a degradable and a non-degradable fraction, so for example VSS = bVSS + nbVSS. The term bVSS is the biodegradable VSS, and nbVSS represents the nonbiodegradable VSS. The nbVSS is the VSS remaining after several years or more of degradation. If the biodegradable portion of the VSS is indeed cellulose and hemicellulose, the composition is approximated as $(C_6H_{10}O_5)_n$. The theoretical oxygen demand of this sugar may be represented as:

$$C_6H_{10}O_5 + 6\ O_2 \rightarrow 6\ CO_2 + 5\ H_2O$$

Hemicellulose, also a polymer of sugars, should have a similar oxygen demand. Given that the molecular weight of the cellulose unit is 162 and O_2 is 32, each gram of cellulose requires 1.18 grams of oxygen to fully oxidize. Assuming all bVSS is cellulose or hemicellulose, the ratio of theoretical BOD to bVSS is approximately 1.2 g BOD/g bVSS.

If the biodegradable portion of VSS is cellulose and hemicelluloses, then the non-biodegradable portion of the organic matter, nbVSS, is lignin. The lignin content of plants varies not only amongst genus and species, but also within a given tree by height. Lignin comprises between 15 and 36% of woody plant mass (Zobel and van Buijyenen 1989). One might thus estimate the bVSS of the plant matter as 65-85% of the total VSS.

Taken together, these estimates of content and BOD may be used to predict the BOD realized after some years of degradation. For example, 1 kg of wood with a 12% water content (a typical value form Desch and Dinwoodie 1996) has 880 g of VSS. If the wood has a 25% lignin content, the cellulose + hemicellulose mass (bVSS) is 660 g. The ultimate BOD is 779 g (= 660 g bVSS * 1.18 g O_2/g bVSS). In a BCR, a fraction of that BOD realized would occur as a result of anaerobic sulfate reduction rather than oxygen respiration.

A.2.2.1 Aerobic Plant Matter Degradation Rates

Before anaerobic wood decomposition can occur to generate 'food' for sulfate reduction, oxygen in the water must first be removed. For most BCRs, the removal mechanism is wood rot, with polysaccharide broken into component sugars and then consumed with oxygen use:

$$\text{Plant matter} \text{ –(cellulose degrader)-> sugar} + O_2 \text{ –(cellulose degrader)-> } CO_2 + H_2O$$

Aerobic cellulose degraders are believed to have much higher rates of wood decomposition than anaerobic.

A.2.2.2 Anaerobic Plant Matter Degradation Rates

The BCR food chain leading to sulfide production is thought of as:

Plant matter –(cellulose degrader) → soluble organics + CO_2 + H_2O + SO_4^{2-} –(SRB)→ CO_2 + H_2O + H_2S

This anaerobic biodegradation of polysaccharides results in soluble organics but is slow.

To determine the rate of sulfide production assuming that this rate is limited by the organic matter supply, one would need to know the rate of creation of the soluble organic, the rate of use of that soluble organic by sulfate reducing organisms, and the stoichiometry by which the organic usage becomes sulfide generation. There have been a few attempts to reveal these rates. For example, plant matter degradation in a BCR was assumed to produce lactate and described using a somewhat complex set of equations (Hemsi et al. 2005). The choice of lactate is somewhat surprising, as few cellulolytic anaerobes produce lactate; ethanol, H_2, succinate and propionate are more common fermentative products (Lynd et al. 2002). The net lactate-producing reaction was given as:

$$2(C_6H_{10}O_5) + 0.427\ H_2O + 0.524\ NH_4^+ \rightarrow 3.126\ C_3H_5O_3^- + 3.652\ H^+ + 0.524\ C_5H_7O_2N$$

where the last term, $C_5H_7O_2N$, is the new-formed cellulose degrader biomass. The lactate is then used by SRBs to form acetate in the stoichiometric ratio, given as:

$$2.128\ C_3H_5O_3^- + 0.0766\ NH_4^+ + SO_4^{2-} \rightarrow 2HCO_3^- + 0.234\ H_2O + H_2S + 2C_2H_3O_2^- + 0.524\ C_5H_7O_2N$$

where the last term is the newly-formed SRBmass. Sulfate reducing bacteria, which fully degrade lactate, are rare compared to the above oxidation to acetate.

Estimating the first step of the process, the rate of woody material degradation in a BCR, is challenging. Limited information is available on rates of degradation in BCRs or BCR-like environments. Also it is important to remember that each BCR has its own chemistry and environmental setting which will affect the observed rate. For example, low pH increases cellulose solubility, thus potentially increasing the rate of degradation, but pH below 5 may severely inhibit cellulose degraders and thereby prevent degradation. Temperature can be a large factor as well, with a common rule of thumb in biology that rates approximately double for every 10° C increase in temperature within the range of an organism's tolerance. Finally, the availability of cellulose for degradation varies tremendously amongst plant species and rate is a function of the amount of area available to cellulose degraders. With those caveats noted, the literature has described wood degradation using first-order or Monod-like kinetics. First-order kinetics are:

$$\frac{dVSS}{dt} = k\left(VSS - nbVSS\right)$$

where:

k = rate constant for VSS degradation, yr^{-1}

Monod-like kinetics are:

$$\frac{dVSS}{dt} = \frac{k_m X (VSS - nbVSS)}{K_x + (VSS - nbVSS)}$$

where:

k_m = maximum specific rate constant for VSS degradation, mg VSS/L/d

X = biomass density of bVSS degraders

K_S = half saturation constant, mg VSS/L

Fitch et al. (2008) applied this Monod model to wetland substrate used in lab-scale BCRs over 7 years, and reported the values for kX of 0.098–1.8 mg VSS/mg VSS-d, and for K, 6.8–210 mg VSS/mg VSS. The values were all normalized to 1 g of BCR substrate, with an observed nbVSS/VSS in the range of 10-13% for a primarily oak chip bark mixture. The variance in the reported Monod-like rate constants is substantial, reflecting the reported rates (from 0.04 mg VSS/g VSS-d to 0.6 mg VSS/g VSS-d).

Hemsi et al. (2005) applied Contois kinetics, a Monod-like approach for the case in which increasing cell mass inhibits the rate of growth but which also describes the case when there is limited surface area of the food (for example, wood particles):

$$\frac{dVSS}{dt} = \frac{k_c X_c (VSS - nbVSS)}{k_c X_c + (VSS - nbVSS)}$$

where, with the reported range of values:

k_c = specific rate of cellulose degradation, 0.625 – 1.25 g/g-d

X_c = biomass density of cellulose degraders 0.0005 – 0.01 times bVSS, which gave values of 9 – 223 mg/L

K_c = Contois saturation constant, 7.5 – 37.5 g/g.

The combination of these values resulted in estimated cellulose decomposition on the order of tens of mg VSS/L-d.

A first order approach aimed at showing the rate of VFA production in landfill-like situations for office paper and cardboard used the first-order approach to model data obtained at 35° C (Qu et al. 2005):

$$\frac{dVFA}{dt} = x_{VFA} k_h I_h W$$

where:

VFA = concentration of VFA in solution, g/L

x_{VFA} = stoichiometric coefficient for cellulose conversion to VFA, 0.66 g/g

kh = first-order hydrolysis rate constant, 0.012 d-1

I_h = inhibition due to pH, I_h = 1 if pH ≥ 7, else:

$$I_h = \exp\left\{-3\left[\frac{pH-7}{7-5}\right]^2\right\}$$

which predicts no hydrolysis at pH 5, and 47% inhibition at pH 6

W = concentration of cellulose in plant material, with an assumption that W = VSS for office paper (mostly cellulose) but W = 0.62 VSS for cardboard because of the higher lignin content, g/L

Comparing these various rate models in terms of predicted VSS use rate, as shown in Table A-2, does not build confidence that a single value may be used for design.

Table A-2. Comparison of predicted rates of anaerobic woody matter degradation

Approach and Reference	Predicted Initial BVSS Use Rate* (mg/L-d)	Observed Rate (mg/L-d)
Contois (Hemsi et al. 2005)	14	nd
Monod-like for down flow (Fitch et al. 2008)	192	13 – 201**
First-order (Qu et al. 2009)	4392	nd
First-order (Wang et al. 2011)	1.8 – 21.5	nd
First-order (Rutkowski et al. 2010)	nd	40
First-order, corn stover (Ruiz et al. 2008)	305	330
First-order, 70% walnut shell, 25% corn stover (Ruiz et al. 2008)	56	32
* assuming BCR substrate is 50 vol%l woody mass (estimated density 30 lb/ft^3) with 30% lignin giving a bVSS density of 336 g/L and using middle of reported value ranges ** normalized to above case, 336 g bVSS/L. nd: No data.		

In addition to kinetic studies noted above, a number of relevant observations have been made without fitting kinetic values. For example, barley straw resulted in more sulfide produced in pore water after eight weeks of flow than did timothy hay or compost (Rabenhorst et al. 1992). Similarly, sawdust was observed to be less degradable than straw when examined as a bulking agent for compost (Zhao et al. 2011) with an oxygen uptake rate (OUR) for straw reported as 2.14 mg O_2/g VS/h at 35° C, and for sawdust, 0.29 mg O_2/g VS/h. Those rates are similar to values determined from anaerobic landfill simulations; for example Wang et al. (2011) reported first-order decay rates based on methane production: inoculu pine, 21.5 yr^{-1}; spruce, 1.8 yr^{-1}; red oak, 2.3 yr^{-1}; and eucalyptus decay was effectively zero. The eucalyptus was found to inhibit conversion of organics to methane.

Decommissioning of a pilot-scale BCR by Golder (Rutkowski et al. 2010) reported a decrease in TOC from 23.2% to 14.4% over 27 months. This corresponds to a first-order rate constant of 0.21 yr^{-1}. A similar order of magnitude rate constant, 0.41 yr^{-1}was observed for corn stover over 13

months in situ, with 36% loss of organic carbon (Ruiz et al. 2008). Over nine months, a mixture of 70% walnut shell and 25% corn stover was fitted by a first order rate constant, 0.061 yr^{-1}, with only 5.4% loss of OC over 9 months, which may have been entirely attributable to the corn stover.

Coastal sediment is somewhat similar to BCR substrate and has been studied in terms of biogeochemical cycle. Canfield et al. (1993) showed carbon oxidation rates to be in the range of 20–105 $\mu mol/cm^3/yr$; assuming a formula of $CH_2O + O_2 \rightarrow CO_2 + H_2O$ as the measured carbon oxidation, this range would be 1.6–8.6 mg/L-d. Corresponding rates of sulfate reduction were 20 – 55 $\mu mol/cm^3/yr$ (55–150 mmol S/m^3-d)

A.3 Sulfate Reduction Rate

Using sulfate as an electron acceptor requires a substantial energy investment rather than yielding energy. It is thermodynamically favorable only at Eh below about -200 mV, which is why oxygen, nitrate, ferric iron and some fermentations occur before sulfate reduction is observed in a system. Sulfate reduction competes with methanogenesis to come in last as a useful compound to which organisms transfer electrons. However, in the absence of these higher energy electron acceptors and with electron donor ("food", generally organic) present, sulfate reducing bacteria (SRB) produce sulfide at some rate.

Most BCRs receive ample sulfate. Therefore the SRB are limited for electron donor. That electron donor is generated by bVSS degradation, so the rate of bVSS degradation will determine the rate at which sulfide is produced. The stoichiometry of that conversion is not completely clear because anaerobic cellulose degraders may produce succinate, propionate, ethanol, or other compounds. Written as reactions to completion, the sulfide stoichiometry varies:

Succinic acid:

$$C_6H_{10}O_5 + 2\ CO_2 + 4\ H^+ \rightarrow 2\ C_4H_6O_4 + H_2O$$

$$2\ C_4H_6O_4 + 7/2\ SO_4^{2-} \rightarrow 8\ CO_2 + 6\ H_2O + 7/2\ S^{2-}$$

Propionic acid:

$$C_6H_{10}O_5 + 4\ H^+ \rightarrow 2\ C_3H_5OOH + H_2O$$

$$2\ C_3H_5OOH + 7/2\ SO_4^{2-} \rightarrow 6\ CO_2 + 6\ H_2O + 7/2\ S^{2-}$$

Ethanol:

$$C_6H_{10}O_5 + 12\ H^+ \rightarrow 3\ C_2H_5OH + 2\ H_2O$$

$$3\ C_2H_5OH + 9/2\ SO_4^{2-} \rightarrow 6\ CO_2 + 9\ H_2O + 9/2\ S^{2-}$$

But the SRBs instead may only partially ferment the organic material, resulting in products such as acetic acid:

Propionate to acetate:

$$C_6H_{10}O_5 + 4\ H^+ \rightarrow 2\ C_3H_5OOH + H_2O$$

$$2\ C_3H_5OOH + 2\ SO_4^{2-} \rightarrow 2\ CH_3COOH + 2\ CO_2 + 2\ H_2O + 2\ S^{2-}$$

Considering the range of sulfide yields shown above, one mole of the glucose-like cellulose unit, $C_6H_{10}O_5$, has a mass of 162 g, and will result in the stoichiometric formation of 64 – 144 g of sulfide. This amount does not consider the formation of cells from the organic will result in a somewhat lower sulfide yield. Assuming the analysis of Hemsi et al. (2005) broadly applies to all such reactions, that about half the carbon goes to cell mass, then 2 x 162 g of organic mass results in 64 – 144 g sulfide evolved, for a yield of 0.20 – 0.44 g sulfide. This amount of evolved sulfide is the equivalent of 6 – 14 mmol S per g VSS consumed. Given a rate of VSS consumption of 100 g/m^3-d, a mid-range value in Table 1-3, the rate of sulfide generation predicted from stoichiometry is 600 – 1400 mmol S/m^3-d. This value is surprisingly close to observed values considering the string of approximations required to arrive at a value.

The stoichiometric approach might allow the designer to estimate the sulfide generation rate and to determine how that rate will decline as the organic is depleted. More data would be required for the specific plant matter used and to estimate the change in degradation with time, as no rigorous long-term study has shown values for such degradation rates.

APPENDIX B. CASE STUDIES

The case studies included in this appendix describe sites where BCRs have been used for remediation. References are included in each case study for more information.

Table B-1. BCR case studies summary

Case Study	Scale	Contaminants	Substrate/Carbon Source	Influent pH	Size/Volume
B.1 Beaver Creek, Ottowa County, OK	Full (planned)	Fe, Zn, Cd, Pb, As	Mushroom compost, hardwood chips, limestone sand, gravel	6.62	≈ 25 gpm (seasonal)
B.2 Mayer Ranch, Ottawa County, OK	Full	Fe, Zn, Ni, As, Pb, Cd	Mushroom compost, hardwood chips, limestone sand, gravel	5.95	150–250 gpm
B.3 Haile Mine Site, Lancaster County, SC	Full	Fe, Cu, ZN	Chipped wood, manure, agricultural limestone, alfalfa	2.0–4.9	6 gpm
B.4 Ferris-Haggerty Mine, Encampment, WY	Pilot	Cu,	Sawdust, alfalfa, limestone, manure, gypsum	Neutral	3– 5 gpm
B.5 Fran Coal Mine, Clinton County, PA	Bench	Al, Co, Cd, Cu, Fe, Zn,	Wood chips, sawdust, limestone, alfalfa, cement kiln dust, manure	2.3	40 gallon drums
B.6 Brewer Gold Mine, Chesterfield County, SC	Full	Cu. Zn, Fe, Mn, Al	Turkey manure, sawdust, phosphate rock, cow manure	3.11	5 gpm
B.7 West Fork Mine, Reynolds County, MO	Full	Pb, Zn, Cd, Cu,	Cow manure, sawdust, limestone, alfalfa	Neutral	1200 gpm
B.8 Leviathan Mine, Alpine County, CA	Full	Al, Cu, Fe, Ni	Ethanol	4.0	30 gpm
B.9 Wheal Jane Mine Cornwall, United Kingdom		Al, Cu, Fe, Mn, Pb, Zn, As	Sawdust, alfalfa, cow manure, limestone	3.0	9 gpm
B.10 Peerless Jenny King Mine, Helena, MT	Full	Zn, Fe, Al, Cu, Ni, Cs	Shredded wood, cow manure	5.7–6.8	5–10 gpm
B.11 Golinsky Mine Site, Shasta County, CA	Full	Fe, Al, Cu, Cd, Mn	Rice hulls, wood chips, alfalfa, limestone, manure	2.5–4.0	10 gpm
B.12 Uranium Remediation Site, Sachsen, Ger-	Bench	Uranium	Limestone	8.0	100 Liter column

Case Study	Scale	Contaminants	Substrate/Carbon Source	Influent pH	Size/Volume
many					
B.13 Copper Basin Mine Site, Polk County, TN	Full	Fe, Mn, Cu, Zn, Al	Alfalfa, mushroom compost, limestone	4.2	≈ 300 gpm
B.14 Lady Leith Mine Case Study, Jefferson County, MT	Full	Fe, Zn	Chipped wood, limestone sand, bacterial inoculum, alfalfa	6.6	6 gpm
B.15 Luttrell Site Case Study	Full	Al, As, Co, Cr, Cu, Fe, Mn, Ni, Pb, Zn	Manure, wood chips, straw, limestone gravel	3.76	1–2.5 gpm

The BCRs described in these case studies are located throughout the U.S. and abroad. Figure B-1 shows the geographic distribution of sites in the U.S.

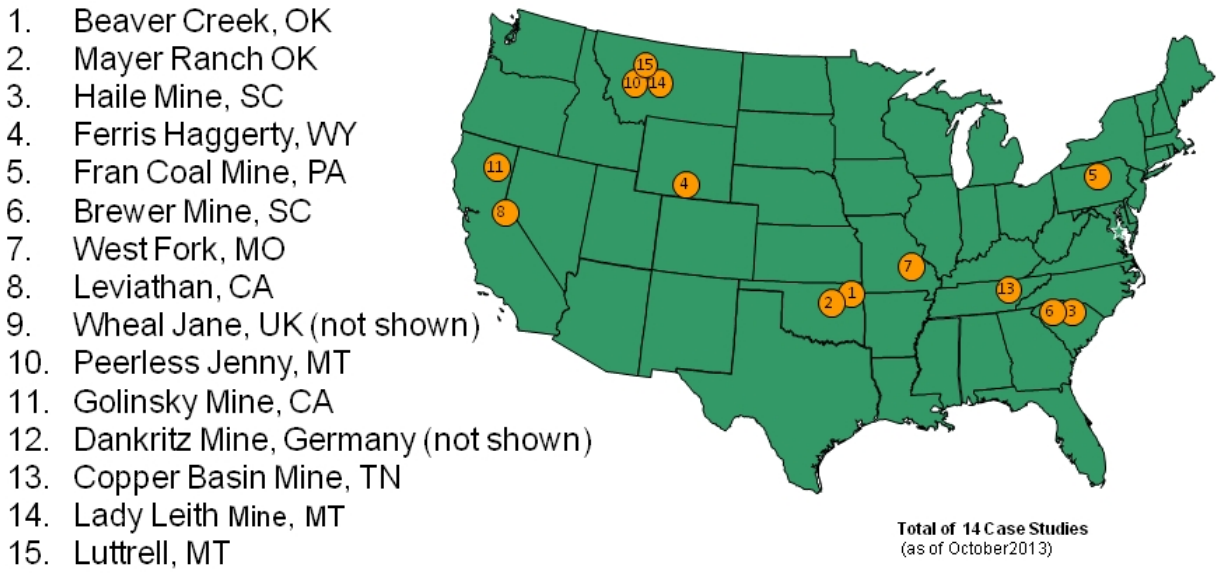


Figure B-1. Geographic distribution of case studies.

B.1 Beaver Creek Mine Discharge Area Case Study

Ottawa County, Oklahoma

Acknowledgments

The mining team would like to acknowledge Tim Kent of the Quapaw Tribe of Oklahoma in Quapaw, Oklahoma who submitted the Biochemical Reactors Case Study.

B.1.1 Site Information

Contact

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tkent@quapawtribe.com

Name, Location, and Site Description:

Beaver Creek mine discharge area, located 1 mile southeast of Quapaw, Oklahoma

Lat/Long: 36° 56' 09" N; 94° 45' 09" W

Elevation: 855 ft.

The Beaver Creek Mine discharge area is near Quapaw, Oklahoma and within the Tar Creek Superfund site, a historical lead and zinc mining area within what is known as the Tri-State Mining District, which covers approximately 2500 square miles in northeast Oklahoma, southeast Kansas and southwest Missouri. Mining began in the Tar Creek area in the early 1900s and continued until the late 1960s using underground room and pillar extraction methods. Galena and sphalerite were the primary lead and zinc ore minerals respectively, mined from Mississippian aged limestone and shale formations. The Tar Creek site was placed on the National Priorities List (NPL) in 1983.

Many of the Mississippian formations mined in the Beaver Creek area, particularly the Boone formation, are aquifers; consequently, the mining operations required continuous dewatering. Once mining ceased in the Beaver Creek watershed in the 1920s, the abandoned mine voids began filling with groundwater, and in some cases surface water, until the mines began discharging water at the surface, and ultimately into Beaver Creek a short time after mine dewatering ceased. Beaver Creek flows through Quapaw Tribal lands that are historically and culturally significant to the tribe.

B.1.2 MIW Chemistry

As mentioned, upon cessation of last mining operations within the Tar Creek site, groundwater began to accumulate in the mine voids. By late 1979, metal-rich waters had begun to discharge, via artesian pressure, into Tar Creek (mine discharges into Beaver creek began much earlier due to the much earlier cessation of mining in this area). These discharges, containing elevated concentrations of iron, zinc, lead, cadmium and arsenic, cause significant degradation of stream water quality. Until 2008, no full-scale treatment of any of these discharges was attempted. In 2008, with grant funding provided by the USEPA Region 6 Water Quality Division to the University of Oklahoma, a full-scale semi-passive treatment system, receiving approximately 150 gallons/minute of contaminated mine water, was constructed to address mine water discharges to an unnamed tributary to Tar Creek, on the Mayer Ranch property near Commerce, OK (approximately 5 miles west of Beaver Creek). This system, consisting of a series of specifically designed process units, has been operational since late 2008 and successfully decreases contaminant concentrations (Nairn et al. 2010a; Figure 1; Table 1).



Figure B.1-1. Oblique aerial photo of existing passive Mayer Ranch treatment system showing multiple process units.

Table B.1-1. Mean water quality changes for existing Mayer Ranch passive treatment system (initial 20 months of operation)

Parameter	In	Out
pH	5.95	7.11
Net alkalinity (mg/L as $CaCO_3$)	29	224

Parameter	In	Out
Iron (mg/L)	192	0.44
Zinc (mg/L)	11	0.45
Cadmium (µg/L)	17	<DL*
Lead (µg/L)	60	<DL
Arsenic (µg/L)	64	<DL
Sulfate (mg/L)	2239	2057
*DL – Detection Limit		

Three monitoring projects in the Beaver Creek watershed (Nairn 2003; Sutter 2008; Nairn et al. 2010b) have identified several mine water discharges (Figure B.1-2) demonstrating considerably lower, yet substantial, contaminant concentrations and discharge rates than those found at the Mayer Ranch property (Table B.1-2). Beaver Creek flows through the tribal pow-wow grounds, as well as through other culturally and historically significant tribal Lands. In-stream surface water quality and biological indicators of ecosystem health are impaired in Beaver Creek (Iverson 2003; Clifton 2009; Nairn 2003; Nairn et al. 2010b). Based on results at the Mayer Ranch BCR system, at-the-source management and/or treatment of these Beaver Creek area discharges would likely result in significant watershed-scale improvements.

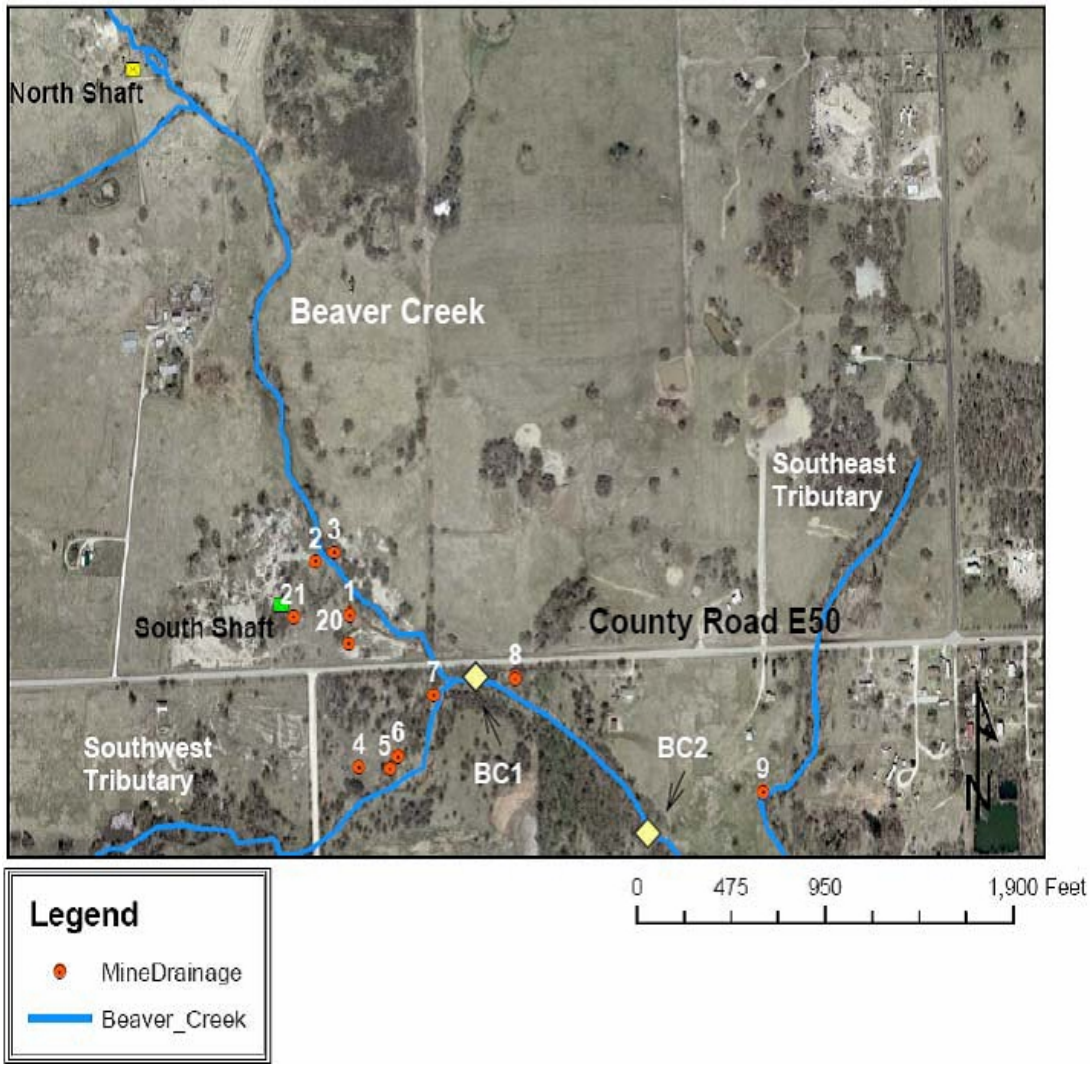


Figure B.1-2. Beaver Creek mine discharge locations.

Aerial photograph of a portion of the Beaver Creek watershed, showing main stem of creek and tributaries, abandoned mine water discharges and open mine shafts. The Quapaw tribal pow-wow grounds are located directly downstream of these contaminated inputs.

Table B.1-2 compares untreated mine water characteristic for existing Mayer Ranch passive treatment system inflow, and Beaver Creek discharge site 8 (see Figure 2 above). Concentrations of contaminants of concern and discharge rates are considerably lower for the Beaver Creek discharge relative to the inflow into the Mayer Ranch property treatment system, while pH and alkalinity are higher, indicating the likelihood of effective semi-passive treatment of Beaver Creek area discharges.

Table B.1-2. Comparison of influent Mayer Ranch vs. effluent Beaver Creek

Parameter	Mayer Ranch PTS Influent	Beaver Creek Site
pH	5.95	6.62
Net alkalinity (mg/L as $CaCO_3$)	29	90
Iron (mg/L)	192	9.28
Zinc (mg/L)	11	1.60
Cadmium (µg/L)	17	1
Lead (µg/L)	60	15
Arsenic (µg/L)	64	20
Sulfate (mg/L)	2239	244
Flow rate (gallons/minute)	150	25

B.1.3 System Design

B.1.3.1 BCR Design

Although the design of the BCR for mine discharges at Beaver Creek is not complete, the design will be similar to that of the Mayer Ranch BCR (multiple process units in series as shown in Figure 3). However, issues specific to the Beaver Creek area MIWs, such as multiple sparsely distributed mine discharge points and intermittent flow characteristics, will have to be addressed during the design phase.

B.1.3.2 Pre and Posttreatment requirements

The Beaver Creek area water treatment system is anticipated to consist of multiple process units, in series, with at least one BCR process unit. Therefore, similar to the Mayer Ranch treatment system, the inflow to the Beaver Creek area treatment system will likely employ some form of oxidation and sediment settling processes before being routed to the BCR unit; and it is anticipated that out-flow from the BCR unit will be routed through a re-aeration pond, a horizontal limestone bed unit, and a final aerobic polishing unit before the water is allowed to discharge at the end of the series of treatment units (Figure 3).

B.1.4 Regulatory Challenges

As mentioned, most tribes have established Environmental Departments funded primarily by USEPA. As with states, USEPA can delegate programmatic authority to tribes for Environmental programs, including regulatory authority under the Clean Water Act, if the tribe applying for such authority has an established Environmental Department and satisfies other USEPA requirements. Consequently, many tribes have developed enforceable Water Quality Standards. Although the Quapaw Tribe has developed its own Water Quality Standards, they are not yet enforceable until the tribe receives programmatic authority from USEPA. However, the tribe has been in close communication with those designing the Beaver Creek area BCR regarding the tribe's Water Quality Standards, which reflect the tribe's priorities in addressing surface water contaminants. Although

the BCR may have the effect of treating water that meets Oklahoma Water Quality Standards, the Quapaw Tribe may not view the resulting water quality as being protective of the tribe, considering the tribe's more intensive use of water and water related resources.

B.1.5 Stakeholder Challenges

The Beaver Creek BCR is proposed to be constructed on Quapaw tribal land that is culturally and historically significant to the tribe. It is referred to as the Catholic 40 property by the tribe. In the late 1800s, the Catholic Church built an Indian school and church on this property so that Quapaw children could attend the school instead of being sent off to far away Indian schools, as was the Bureau of Indian Affairs policy for children of other Native American tribes during this period of Indian Assimilation. There is also a tribal cemetery on this property and members practice traditional subsistence and cultural gathering of plants along Beaver Creek in this area. Finally, Beaver Creek flows through the Quapaw Tribe's pow-wow grounds immediately downstream of the proposed BCR. Tribal members gather various food and ceremonial resources along Beaver Creek during the annual pow-wow and the creek plays an important part in their cultural and religious practices. Consequently, it has been extremely important to seek input from the tribal leaders and the tribe's environmental Department (like States, most tribes now have Environmental programs funded by USEPA) early on in the conceptual design of the BCR and certainly before the initiation of any field investigations.

Most Native American tribes have a different perception of environmental protection and remediation than that of western European or modern American societies. Therefore, it is very important to approach the tribe(s) that may be affected by the proposed BCR, and initiate a consultation process before beginning even the conceptual design. Federally Recognized Tribes and USEPA have jointly developed consultation policies and procedures to be used before environmental regulations or actions, that may affect tribes, are initiated. Therefore, it is highly recommended that those proposing BCRs on, or near, tribal lands or in areas of tribal jurisdiction, contact the appropriate USEPA Regional office and ask for office of the USEPA tribal liaison official who will likely offer detailed guidance on the appropriate tribal consultation procedures.

Regarding the proposed Beaver Creek BCR, the following step-by-step procedure was followed to incorporate tribal input into information and data gathering related to the proposed BCR:

1. Dr. Robert Nairn of Oklahoma University was encouraged by USEPA to contact the Quapaw Tribe Environmental Department to discuss the possibility of beginning data gathering involved in investigating the feasibility of constructing a BCR in on the Catholic 40 property in the Beaver Creek area.
2. Tim Kent, Quapaw Tribe Environmental Director, approached Quapaw Tribal leaders to explain the general concept of a BCR (using the Mayer Ranch BCR as an example) and to inquire about the possibility of collecting the data necessary to adequately evaluate the feasibility of constructing and operating a BCR in the Beaver Creek area.
3. After asking many questions regarding the effects of the BCR and its construction and maintenance, the Quapaw Tribal Business Committee reluctantly approved the initiation of data

gathering and monitoring related to the BCR feasibility study, contingent on the assurance that no historic or culturally significant areas or features would be disturbed during the data collection activities (the Quapaw Tribe, like many native American tribes, believe that man cannot fix Mother Earth once man has severed the sacred connection between the two; therefore, the tribe is very skeptical about the value in attempting the construction of a BCR).

4. Assessment and mapping of historic and culturally significant features on the Catholic 40 property, where the sampling and construction related to the proposed BCR is planned, was undertaken by the tribe's Historic Preservation Officer (THPO).
5. Upon the assurances from the THPO and the tribe's Environmental Director that sampling and construction related to the proposed BCR would not adversely affect any historic or culturally significant features, the Tribal Business Committee granted approval to initiate the design of the BCR. Once a draft of the design is complete, it will be presented for approval to the Tribal Business Committee.

B.1.6 Other Challenges and Lessons Learned

The primary lesson learned during the planning of the Beaver Creek BCR was: when considering the design and construction of a BCR, entities should investigate the likelihood that the BCR would affect tribal lands by contacting the liaison office of the appropriate USEPA Region. If the BCR is planned for an area near, or upstream of, tribal lands, the potentially affected tribe should be consulted using USEPA/tribal guidance.

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B.2 Mayer Ranch/Commerce Site Case Study

Ottawa County, Oklahoma

The mining team would like to acknowledge David Cates, Oklahoma DEQ, and Dr Robert Nairn, University of Oklahoma, for submitting this Biochemical Reactors Case Study.

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Name, Location, and Site Description

The site is located in the Tar Creek Superfund Site, part of the Picher Mining Field, and the northeastern Oklahoma portion of the former lead (Pb) and zinc (Zn) mining area known as the Tri-State Mining District. In this area, the ore deposit consists of Pb and Zn sulfides associated with cherty carbonate host rock. The principal ore host stratum is the Boone Formation, composed of fossiliferous dolomite, limestone and nodular chert. Principal ore minerals are sphalerite and galena, with secondary concentrations of chalcopyrite, enargite, luzonite, marcasite, pyrite, and barite. Significant quantities of Pb and Zn were produced from the Tri-State District from the 1890s through the 1960s. By the late 1950s, depressed global markets resulted in the suspension of most mining operations. By the early 1970s when mining ceased, almost 2 million tons of Pb and 9 million tons of Zn had been produced. The Tar Creek site (Figure 1) was proposed for the Comprehensive Environmental Response, Compensation, and Liability Act (Superfund) National Priorities List (NPL) in 1981 and received final listing in 1983. Nearby are the Cherokee County Superfund Site (Kansas) and the Oronogo-Duenweg Mining Belt and Newton County Mines Superfund Sites (Missouri). The following information contains text taken directly from Nairn et al. (2009).

In the Oklahoma portion of the district, approximately 1,000 hectares (ha) are underlain by underground mines in all or part of 47 sections of the Tar Creek Superfund site. During mining, large-

capacity dewatering operations pumped approximately 50,000 m^3 of water per day from the mines. Upon decline and cessation of mining, groundwater began to accumulate in the mine voids. Approximately 94 million m^3 of contaminated water exist in underground voids. By late 1979, metal-rich waters began to discharge into Tar Creek and its tributaries. The first documented discharge of mine drainage was at a location near southeast Commerce, Oklahoma, denoted by the Oklahoma Water Resources Board as Site 14.

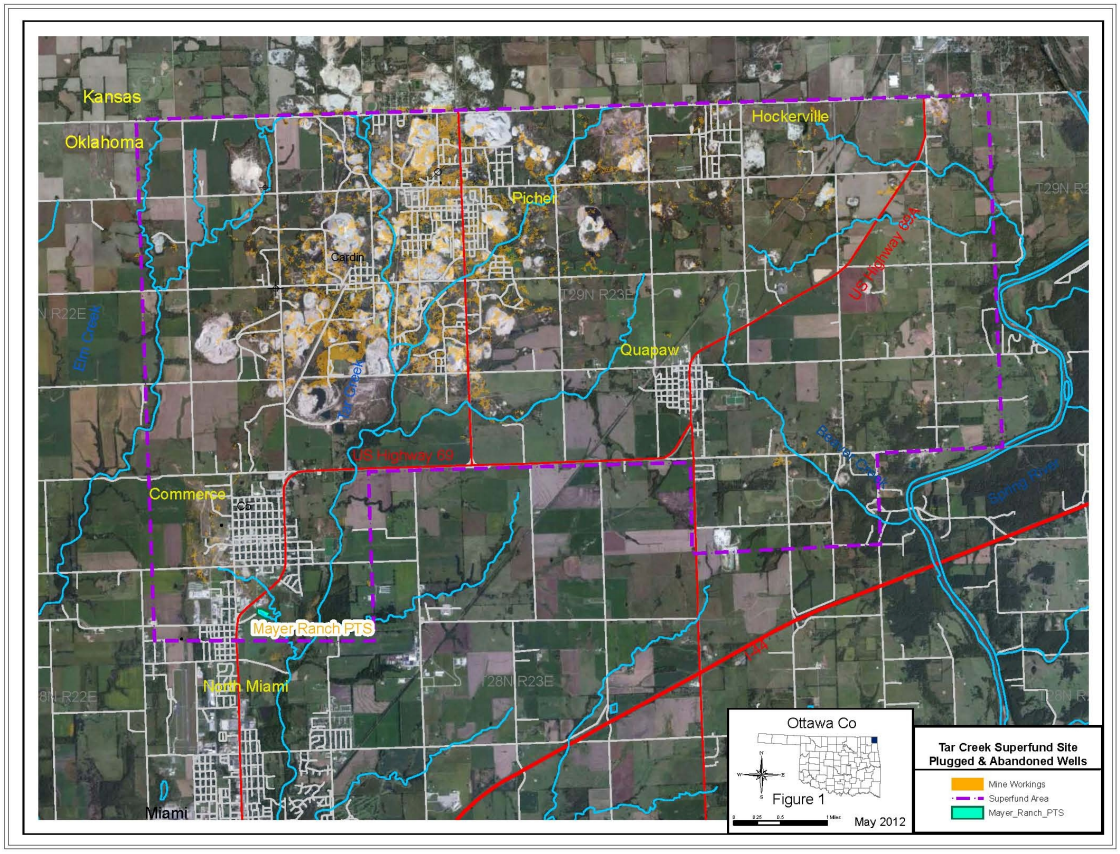


Figure B.2-1. Tar Creek Superfund Site and location of Mayer Ranch PTS.

B.2.1 MIW Chemistry

Artesian groundwater discharges from abandoned underground lead and zinc mines into a tributary of Tar Creek for over 30 years. Sampling of seeps from two old mining boreholes showed concentrations, as high a 192,000 µg/l iron, 11,000 µg/l zinc, 970 µg/l nickel, 64 µg/l arsenic; 60 µg/l lead, and 17 µg/l cadmium from 2004 to 2008. The seeps discharged at a rate (as measured with bucket and stop watch) ranging from 570 to 950 L/min (150 to 250 gpm) with a pH of 5.95. Other trace metals of concern are: aluminum, cobalt, and manganese. The water is net alkaline with CO_2 degassing at the surface. The influent carbonate, acidity, and sulfate concentrations are: total alkalinity 393 mg/L $CaCO_3$, total acidity 364 mg/L $CaCO_3$, and sulfate 2,264 mg/l.



Figure B.2-2. Picture of Mayer Ranch Mine water discharge (OWRB Site 14).

B.2.2 System Design

The U.S. Environmental Protection agency (USEPA), University of Oklahoma, and other federal, state, local and tribal partners collaborated in 2004 to design a PTS on part of the 40-square mile Tar Creek Superfund site in northeast Oklahoma. Through a competitive bidding process, a design/build engineering contract was awarded to CH2M-Hill using funds provided by the U.S. Environmental Protection Agency and U.S. Geological Survey. The intent of this project was to develop engineering plans and specifications for the design and construction of the passive treatment system, build the system, and provide as-built construction documents. The goal was to decrease receiving stream concentrations of iron, zinc, lead, and cadmium to the criterion continuous concentrations suggested by USEPA's national recommended water quality criteria for fresh water aquatic life. Engineering design, based on the conceptual designs and taking into account on-site conditions, was undertaken as an iterative process between the design firm, the University of Oklahoma, and USEPA. To accommodate site constraints, multiple adjustments were made to the system layout. Most notable of these included the configuration of the system as two parallel flow paths to allow estimation of variance in performance and to perform maintenance, consolidation of the layout into a more compact "footprint" area to comply with construction requirements adjacent to an existing utility corridor, and minimize impacts to adjacent landowners, thereby creating a flatter hydraulic profile. Rotosonic overdrilling of both mine drainage seeps was completed 18 months prior to passive treatment system construction, thus providing hydraulic control and allowing additional investigations to further quantify flow rates and variability.

Design and construction tasks included capture and control of the two known artesian mine drainage discharges, diversion of stormwater flows from a 470-ha upgradient watershed, implementation of all passive treatment process units including water conveyance structures, and provision of as-built documents. Design performance was estimated using inflow concentrations of 192 mg/L (Iron) Fe, 11 mg/L Zn, 17 µg/L (Cadmium) Cd and 60 µ/L Pb, and Fe removal rate of 20 grams per square meter per day with target Fe effluent of 1 mg/L and assuming net alkaline conditions, a discharge rate of 1000 L/minute, available site area of approximately 3.6 ha, and treatment area of 2 ha.

Construction began in July 2008 and was completed in late November 2008. Issues requiring resolution during construction and start-up included the incorporation of a third seep (Seep D) into the inflow oxidation pond, removal of debris and existing iron oxide muck, delays throughout the course of construction caused by a record rainfall during the summer and related site water management issues, and changes in organic substrate requirements. The third seep was isolated as a distinct inflow into the first process unit and did not result in inflow discharge rates or mass loadings substantially different from initial designs. Based on additional laboratory work, the vertical-flow bioreactors received a mixture of 45% spent mushroom compost, 45% hardwood chips, and 10% manufactured limestone sand in the organic substrate layer. The total cost of design and construction totaled $1,196,000. The system is expected to last 25 years, with an estimated $20,000 annual operation and maintenance (O&M) cost. Total costs (including research) are estimated at $4 million.

The completed system includes 10 distinct process units with a single initial oxidation pond (cell 1) followed by parallel surface-flow aerobic wetlands/ponds (cells 2N and 2S), vertical-flow bioreactors (cells 3N and 3S), re-aeration ponds (Cells 4N and 4S) and horizontal flow limestone beds (Cells 5N and 5S), and a single polishing pond/wetland (Cell 6). Mine water was diverted into the passive treatment system for the first time on December 2, 2008.

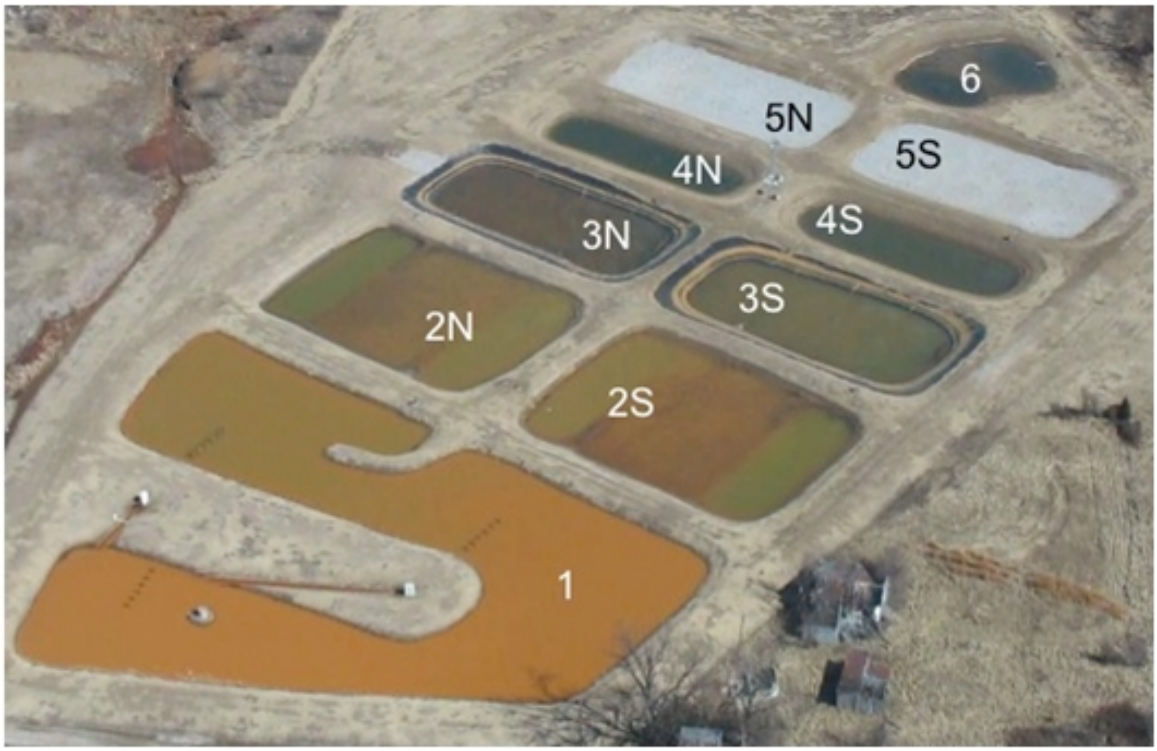


Figure B.2-3. Oblique aerial photograph of passive treatment system taken showing mine drainage discharges and individual process unit designations.

B.2.2.1 BCR Design

Water flows from the aerobic wetlands / ponds to the third process unit comprising parallel vertical flow BCRs designed to provide a reducing environment to remove cadmium, lead and zinc. The BCRs were constructed with a 45-cm (18-inch) layer of organic substrate (a mixture of 45% spent mushroom compost, 45% hardwood chips, and 10% manufactured limestone sand) overlying a 30-cm limestone gravel drainage layer containing perforated drainage pipes. The substrate composition was determined through mainly laboratory testing and was modified slightly prior to installation to increase its hydraulic conductivity. A low permeability HDPE liner beneath the gravel maintains design integrity.

The sizing was designed using the 0.3 moles/m3-d volumetric sulfate reduction rate and the sum of the system influent molar loading of zinc, cadmium, nickel, and lead. As it turns out the system may be over sized designed somewhat since the influent concentration of metals to the BCR is somewhat reduced by pretreatment due to co-precipitation with iron in the oxidation pond and aerobic wetlands. to the Sulfate concentration changes in the vertical-flow bioreactors, although demonstrating a great deal of temporal variability, provided mean calculated sulfate reduction rates of 0.53 and 0.40 mol / m^3 - d for cells 3N and 3S, respectively.

The system and BCR became operation in December 2008. There was no inoculation period as the mine discharges were turned into the system at this time. High H_2S odors were apparent initially

but have reduced over time to the point that they are only noticeable during the summer months. Water samples for biological oxygen demand (BOD) and sulfide analyses are currently being collected at the inlet and outlet of Unit 4 to evaluate the re-aeration process.

B.2.2.2 Pre- and Posttreatment Requirements

The first process unit (Table B.2-1) is an oxidation pond that receives the artesian MIW and oxidizes and hydrolyzes the Fe^{2+}. Oxidized iron then settles out as iron oxyhydroxide solids. Effluent water flows by gravity feed to the two parallel surface water wetlands. Emergent vegetation in the shallow zones (<0.3 m) increases retention time of solids, while the deeper pools facilitate settling of oxidized and hydroloyzed iron solids from solution.

In the fourth process unit submerged aeration systems in two parallel aeration ponds re-oxygenate water from the BCRs. Aeration reduces odors caused by the emission of hydrogen sulfide gas during bacterial sulfate reduction. Water in one pond is aerated by a vertical displacement pump powered by a 20-foot-tall windmill; as a result, water is aerated only when the wind is blowing. Water in the adjacent pond is aerated by a high- volume compressor powered by a 120-watt photovoltaic (PV) system that charges a deep-cycle battery. The PV system operates the compressor on a 20-hours-on, 4-hours-off cycle. The passive use of a solar panel and a windmill requires minimal maintenance and uses a natural process – not fossil fuels.

Horizontal-flow limestone beds in ponds of the fifth process unit further improve water quality by removing additional zinc and manganese and increasing water hardness to offset the bioavailability of any remaining trace metals. The limestone beds are greater than 1 meter thick to provide a minimum 14-hour retention time.

The final aerobic polishing pond / wetland facilitates settling of residual solids through vegetative filtration and re-aerates water through passive photosynthesis. Water from the pond discharges via an 8-inch exit pipe with flow meter to a limestone boulder cascade for aeration, then to a small channel containing a weir for monitoring flow that feeds into the Tar Creek tributary. The total retention time for the complete system is about 3 weeks.

Table B.2-1. Summary of process units, primary targeted water quality parameters and design function for passive treatment system.

Process unit	Targeted parameter	Function
Oxidation pond	Fe	Oxidation, hydrolysis and settling of iron oxyhydroxide solids; Trace metal sorption
Surface-flow wetlands/ponds	Fe	Solids settling
Vertical-flow bioreactors	Zn, Pb, Ni and Cd	Retention of trace metal sulfides via reducing mechanisms
Re-aeration ponds	Oxygen demand and odor	Wind- and solar-powered re-aeration; Oxygen demand and H_2S stripping; Addition of O_2

Process unit	Targeted parameter	Function
Horizontal-flow limestone beds	Zn, Mn and hardness	Final polishing of Zn as $ZnCO_3$ and of Mn as MnO_2; Hardness addition to offset bioavailability of remaining trace metals
Polishing pond/wetland	Residual solids	Solids settling: Photosynthetic oxygenation; Ecological buffering

B.2.3 BCR Performance

In the vertical-flow bioreactors, Zn concentrations decreased from 5.76 ± 0.81 mg/L to 0.24 ± 0.25 mg/L in Cell 3N and from 5.97 ± 0.87 mg/L to 0.28 ± 0.25 mg/L in Cell 3S. These cells were designed specifically to remove Zn and other trace metals through reductive mechanisms, converting free Zn and SO_4^{2-} to ZnS. Removal rates were much higher in these cells (1.46 ± 0.27 and 1.45 ± 0.21 g / m^2- d in cells 3N and 3S, respectively) than in cell 1 (1.46 ± 0.27 and 1.45 ± 0.21 g m-2d-1 in cells 3N and 3S, respectively). Biological sulfate reduction in vertical-flow bioreactors is well-documented as a metal removal mechanism (e.g., Jong and Perry 2006; Neculita et al. 2007) and was the primary Zn retention processes for this passive treatment system.

Although the vertical-flow bioreactors were designed to remove Cd and Pb as well as Zn, Cd and Pb rarely remained in measureable concentrations at this stage of the treatment system. On two instances (January and August 2009), Cd was detected in the inflow to the vertical-flow bioreactors at a concentration of approximately 1 µg/L, but was removed through reductive processes occurring within the cells.

A small percentage (<10%) of Ni was removed through co-precipitation and sorption in Cell 1. However, the majority of Ni (~95%) was removed via reductive mechanisms in the vertical-flow bioreactors. The concentration decreased from 0.81 ± 0.03 mg/L to 0.05 ± 0.04 mg/L in Cell 3N and from 0.82 ± 0.04 mg/L to 0.04 ± 0.03 mg/L in Cell 3S. Ni concentrations did not decrease significantly in the remainder of the system, with final system effluent concentrations of 0.035 ± 0.01 mg/L.

Flow-weighted mean SO_4^{2-} concentrations were elevated in the discharges (2,239 ± 26 mg/L) and decreased to 2,047 ± 72 mg/L in the final outflow. Sulfate concentration changes in the vertical-flow bioreactors, although demonstrating a great deal of temporal variability, provided mean calculated sulfate reduction rates of 0.53 and 0.40 mol / m^3 - d for cells 3N and 3S, respectively. These values are slightly greater than reported values for field-scale systems of approximately 0.3 mol m-3d-1 as reported earlier (e.g., Dvorak et al. 1992; Neculita et al. 2007) but were not unexpected during this initial operational period.

B.2.3.1 Overall Water Quality Changes

The passive treatment system described here demonstrated that the quality of these artesian hard-rock mine discharges may be effectively addressed through an appropriate combination of ecologically engineered process units. Final effluent concentrations of Cd, Pb and As were below

detection limits and percentage decreases for Fe, Zn and Ni in the entire system were 99.7, 98.8 and 96.3%, respectively (Table B.2-2). Conservative ion (e.g., Mg and Li) concentration changes were between 0.3 and 1.2% indicating that design-targeted biogeochemical mechanisms affected contaminant concentrations and dilution played little if any role.

Table B.2-2. Summary total metals and sulfate concentration data for flow-weighted influent and final system effluent. Data are presented in mg/L as mean ± standard error; BDL = below detection limits.

Parameter	Influent	Final Effluent	Removal
Al	0.094 ± 0.009	0.071 ± 0.030	24.47
As	0.063 ± 0.002	BDL (0.022)	65.08
Ca	742 ± 9.0	740 ± 22.3	
Cd	0.016 ± 0.002	BDL(0.00064)	96.0
Co	0.066 ± 0.008	0.007 ± 0.0004	89.39
Cr	0.001 ± 0.0002	0.002 ± 0.0006	
Cu	0.002 ± 0.0003	0.003 ± 0.0003	
Fe	177 ± 2.33	0.57 ± 0.207	99.7
K	26.0 ± 0.286	31.1 ± 4.82	
Li	0.366 ± 0.010	0.365 ± 0.018	
Mg	200 ± 2.53	198 ± 7.49	
Mn	1.51 ± 0.016	1.38 ± 0.197	8.61
Na	94.9 ± 1.63	96.6 ± 4.23	
Ni	0.945 ± 0.015	0.035 ± 0.007	96.30
Pb	0.068 ± 0.003	BDL(0.019)	72.06
Zn	8.29 ± 0.078	0.096 ± 0.037	98.84
SO_4	2239 ± 26	2047 ± 72	

The PTS retains approximately 57,000 kg (>62 tons) iron, 3,300 kg (4 tons) zinc, 300 kg (660 lbs) nickel, 100 kg (200 lbs) aluminum, 18 kg (40 lbs) arsenic, 17 kg (37.5 lbs) lead, and 5 kg (11 lbs) of cadmium per year. Beneficial reuse of the accumulated solids and substrates retained by the system is being explored as researchers continue to study receiving stream biogeochemistry, fish and macro-invertebrate communities, microbial community activity and degree of bioaccumulation potential.

B.2.4 System Performance

In the first two years of operation, the passive treatment system performed as designed from a water quality perspective. The net alkaline nature of the mine waters was maintained throughout the system as target metals were removed from solution. Due in part to degassing of elevated CO_2 concentrations in the artesian discharges, pH increased from less than 6 to >7 at final discharge. Total Fe concentrations decreased to 0.57 ± 0.59 mg/L at the final outflow with the great majority of total iron removal in the initial oxidation pond. Total Zn concentrations decreased to 0.10 ± 0.10

mg/L for the entire system, with removal occurring via sorption/co-precipitation in cell 1, sulfide precipitation in cells 3N and 3S, and $ZnCO_3$ formation in cells 5N and 5S. Total Cd, Pb, and As concentrations were decreased to below detection limits (0.64, 19.5 and 22 µg/L, respectively) before the outflow of the second process units, presumably through sorptive mechanisms. The majority of Ni was removed via reductive mechanisms in the vertical-flow bioreactors, with final system effluent concentrations of 0.035 ± 0.007 mg/L.

Coupled oxidative and reductive processes lead to fluctuating dissolved oxygen (DO) and ORP values throughout the system and a demonstrated seasonality as the rates of biological processes varied. The remaining discussion focuses on constituents of interest.

B.2.4.1 Iron Retention

During this initial period of performance, flow-weighted mean total Fe concentrations decreased from 177 ± 2.33 mg/L to 0.57 ± 0.21 mg/L at the final outflow (Table 3). The great majority of total iron removal occurred in the initial oxidation pond and parallel surface-flow aerobic wetlands/ponds, with mean removal rates of 21 ± 2.8 and 3.7 ± 3.4 g / m^2-d, with the latter rate being load-limited. The initial oxidation pond was designed specifically for iron oxidation, hydrolysis, precipitation, and settling. A statistically significant relationship between area-adjusted iron removal rate and effluent water temperature did exist for this process unit ($r = -0.83$, $p < 0.01$) indicating greater removal under colder conditions.

The parallel surface-flow aerobic wetlands/ponds were designed for iron oxide solids retention and include influent and effluent deep water areas with an interior shallow wetland shelf. Although planted, vegetation establishment on this wetland shelf was limited in this first growing season (<10% coverage), perhaps inhibiting solids retention.

B.2.4.2 Zinc Retention

Flow-weighted mean total Zn concentrations decreased from 8.29 ± 0.08 mg/L to 0.10 ± 0.04 mg/L for the entire system. Zinc was targeted for retention via several mechanisms in individual process units of the treatment system, including sorption (Cell 1), ZnS precipitation (Cells 3N and 3S), and $ZnCO_3$ formation (Cells 5N and 5S). Over the course of initial operation, Zn was removed in all of these cells. In the initial oxidation pond, Zn concentrations decreased to 6.71 ± 0.37 mg/L, representing an area-adjusted removal rate of 0.28 ± 0.05 g/m^2d. Retention likely occurred through co-precipitation and sorption to iron oxyhydroxide precipitates.

In the vertical-flow bioreactors, Zn concentrations decreased from 5.76 ± 0.81 mg/L to 0.24 ± 0.25 mg/L in Cell 3N and from 5.97 ± 0.87 mg/L to 0.28 ± 0.25 mg/L in Cell 3S. These cells were designed specifically to remove Zn and other trace metals through reductive mechanisms, converting free Zn and SO_4^{2-} to ZnS. Removal rates were much higher in these cells (1.46 ± 0.27 and 1.45 ± 0.21 g / m^2d in cells 3N and 3S, respectively) than in cell 1. Biological sulfate reduction in vertical-flow bioreactors is well-documented as a metal removal mechanism and was the primary Zn retention processes for this passive treatment system.

Table B.2-3. Selected mean water quality data from artesian discharges and process unit effluents. All data are mg/L except pH (standard units) and alkalinity (mg/L as $CaCO_3$). BDL = below detection limits.

		pH	DO	Alk	Fe	Zn	Cd	Pb	Mn	As	Ni	SO_4
Discharges	SA	6.0-1	1.1-7	402	171	8.34	0.01-5	0.06-5	1.4-7	0.06-2	0.94	220-9
	SB	5.9-7	1.0-4	371	178	8.15	0.01-6	0.06-9	1.5-1	0.06-4	0.94	226-4
	S-D	5.9-9	1.2-7	386	181	9.00	0.01-5	0.07-0	1.6-2	0.06-1	0.96	234-3
Cell 1		6.1-0	3.8-3	161	30	6.71	0.00-3	0.02-9	1.4-8	BDL	0.87	217-3
Cell 2	N	6.5-1	7.9-4	135	4.4	5.76	BDL	BDL	1.5-0	BDL	0.81	219-9
	S	6.4-0	7.0-0	131	4.4	5.97	BDL	BDL	1.5-8	BDL	0.82	221-6
Cell 3	N	6.8-5	0.3-5	262	1.3	0.24	BDL	BDL	1.3-6	BDL	0.04-5	231-1
	S	6.8-4	0.2-8	242	1.1	0.28	BDL	BDL	1.3-5	BDL	0.04-2	209-1
Cell 4	N	7.2-0	3.8-6	236	0.8-7	0.16	BDL	BDL	1.3-8	BDL	0.03-5	207-6
	S	7.1-0	3.5-1	232	0.9-2	0.23	BDL	BDL	1.4-4	BDL	0.04-2	209-9
Cell 5	N	6.9-8	0.6-5	229	0.5-7	0.08-7	BDL	BDL	1.2-8	BDL	0.03-3	206-4
	S	6.9-5	0.4-2	227	0.6-3	0.10-9	BDL	BDL	1.4-0	BDL	0.03-3	219-5
Cell 6		7.1-1	2.6-5	224	0.5-7	0.09-6	BDL	BDL	1.3-8	BDL	0.03-5	205-7

In the horizontal-flow limestone beds, Zn concentrations decreased from 0.16 ± 0.21 mg/L to 0.09 ± 0.11 mg/L in Cell 5N and from 0.23 ± 0.23 mg/L to 0.11 ± 0.12 mg/L in Cell 5S. Relatively low influent Zn concentrations resulted in load-limited removal rates of 0.03 ± 0.04 and 0.02 ± 0.02 g / m^2d for cells 5N and 5S, respectively. These cells were designed to remove Zn through mechanisms specific to proper pE-pH. Although pE values (converted from field ORP readings) were typically within the needed range for amorphous $ZnCO_3$ and ZnO formation (-0.1 to +0.9 v), pH values were 6.95-7.20, below the needed pH values of 7.5-8.2.

B.2.4.3 Trace Metals

Other metals of specific interest in these waters were Cd, Pb, and As. All three were removed to below detection limits (0.64, 19.5 and 22 µg/L, respectively) before the outflow of the second process units, presumably through sorptive processes. Although the vertical-flow bioreactors were

designed to remove Cd and Pb as well as Zn, Cd and Pb rarely remained in measureable concentrations at this stage of the treatment system. On two instances (January and August 2009), Cd was detected in the inflow to the vertical-flow bioreactors at a concentration of approximately 1 µg/L, but was removed through reductive processes occurring within the cells.

The other trace metal found in significant concentrations in these waters was Ni. A small percentage (<10%) of Ni was removed through co-precipitation and sorption in Cell 1. However, the majority of Ni (~95%) was removed via reductive mechanisms in the vertical-flow bioreactors. The concentration decreased from 0.81 ± 0.03 mg/L to 0.05 ± 0.04 mg/L in Cell 3N and from 0.82 ± 0.04 mg/L to 0.04 ± 0.03 mg/L in Cell 3S. Ni concentrations did not decrease significantly in the remainder of the system, with final system effluent concentrations of 0.035 ± 0.01 mg/L.



Figure B.2-4. Clean water exiting the Mayer Ranch Treatment System (Pond 6).

B.2.4.4 Sulfate

Flow-weighted mean SO_4^{2-} concentrations were elevated in the discharges (2,239 ± 26 mg/L) and decreased to 2,047 ± 72 mg/L in the final outflow. Sulfate concentration changes in the vertical-flow bioreactors, although demonstrating a great deal of temporal variability, provided mean calculated sulfate reduction rates of 0.53 and 0.40 mol m-3d-1 for cells 3N and 3S, respectively. These values are slightly greater than reported values for field-scale systems of approximately 0.3 mol/m^3 - d as reported earlier (e.g., Dvorak et al. 1992; Neculita et al. 2007) but were not unexpected during this initial operational period.

B.2.4.5 Receiving Stream

Biological testing in the receiving stream down-stream from the BCR system shows the presence of less tolerant species of fish that have been absent from the Tar Creek tributary for more than

thirty years - since the mine water discharge began in 1979. These species include darters, sunfish, and large-mouth bass.

B.2.5 BCR Monitoring

Future research at this site will focus on development of a detailed mechanistic understanding of passive biogeochemical processes leading to water quality changes. Research infrastructure has been installed to examine sedimentation rates in the initial oxidation pond, pore water chemical composition in the vertical-flow bioreactors, oxygenation kinetics in the re-aeration cells, dissolution/precipitation dynamics in the horizontal-flow limestone beds, and evaluation of metals transport after high flow rainfall events by means of auto-samplers at the inlet and outlet of the aerobic wetlands and the system outlet. Ancillary ecological benefits provided in the passive treatment system itself and in the receiving waters will be examined, as well as public educational opportunities.

Monthly monitoring for the first three years of system operation has transitioned to quarterly monitoring. This level of monitoring consists primarily of collection of water samples (for analyses of seven metals, alkalinity, and sulfate, as well as field parameters: dissolved oxygen, specific conductance, pH and temperature) at the inflow and outflow of each process unit and measurement of volumetric flows (with bucket and stop watch at the system influent and a weir and flow meter with pressure transducer at the effluent). Some continuous data are collected from YSIs and data loggers installed at the system influent and outfall. This stored data (dissolved oxygen, specific conductance, pH and temperature) is downloaded during these monitoring events. Other research data is collected during these monitoring events. For example, water samples at the inlets and outlets of the re-aeration ponds for BOD and sulfide analyses are now collected monthly to evaluate the kinetics of oxygenation in the re-aeration ponds. Also auto-samplers stationed at the outlets of the oxidation pond, the aerobic wetlands, and the system outlet (polishing pond) will be used to collect samples after high flow rainfall events to evaluate metals re-mobilization and transport through the system.

The YSI multi-meters installed at continuous monitoring sites are inspected during periodic monitoring and are serviced annually. An optical dissolved oxygen probe with a wiper is now used on the YSI multi-meter since the membrane probes tended to foul often from iron oxyhydroxide precipitates.



Figure B.2-5. Inspection of a YSI multi-meter.

Routine maintenance was conducted during the monthly monitoring events. Currently this is being accomplished by OU during the quarterly monitoring and weekly by interns from the local college (Northeastern Oklahoma A&M) hired under a cooperative agreement with the City of Commerce, the Quapaw Tribe of Oklahoma, and the University of Oklahoma. The weekly duties involve mowing, keeping the inlets and Agra-drains clear of vegetation, visual inspection for normal operations and reporting problems. They also collect samples from the auto samplers and mail them to OU.

BCR longevity is evaluated through periodic coring of the substrate. So far no reaction front has been observed. The design life of the BCR is for 30 years.

The area is filled with wildlife, including golden winged butterflies, turtles, frogs, muskrats, redwing blackbirds, ducks and a few geese. A wildlife camera has recorded other animals including deer and bobcat. No vandalism has occurred even though trespassers have been seen on the wildlife camera (including kids fishing in Pond 6).

B.2.6 Regulatory Challenges

This passive treatment system represents the first full-scale mine water treatment system in the mining district. At this time, the fund-balancing waiver regarding surface waters under the OU 1 ROD is still in place and any concerted effort to address water quality has not been undertaken. However, in the Fourth Five Year review of the OU 1 ROD, it was noted that passive treatment "could be an economically feasible engineered remedy for surface water at the site. For these reasons, in this fourth five-year review, the fund balancing ARARs waiver included in the OU1 ROD may no longer be appropriate and should be reevaluated" (USEPA 2010).

The discharges addressed by the passive treatment system represent approximately 20% of the contaminant mass loading from artesian mine drainage discharges in the Tar Creek watershed. The stream and its tributaries receive additional contaminant mass loading from other artesian discharges, waste pile and pond runoff, and other sources. It is obvious that considerable efforts would be required to satisfactorily address mining-related water quality deterioration in this watershed. In 1994, approximately 15 years after the first discharges began to flow, the State of Oklahoma concluded that "the impacts to Tar Creek… are because of irreversible man-made conditions" and furthermore USEPA "concurs with the State's conclusion that the surface water conditions are irreversible" (USEPA 1994). However, appropriately designed passive treatment systems appear to be capable of treating these waters effectively and evaluation of future watershed-scale applications is warranted.

From the bioavailability and eco-toxicity perspectives, the untreated artesian discharges exceeded the hardness-adjusted USEPA Criterion Continuous Concentrations (CCC) for Cd, Fe, Ni, and Zn and the Criterion Maximum Concentration (CMC) for Zn (USEPA 2010b). In general, as hardness increases the toxicity of the metal decreases. Despite the fact that there was no significant increase in hardness values in the system, the system effluent no longer exceeds any of these criteria. LaBar et al. (2010) report on stream water quality changes related to operation of this passive treatment system.

The Mayer Ranch pilot project was conducted within a Superfund site under the oversight of the USEPA and consequently a discharge permit was not required. Also mitigation of the volunteer (cattail) wetlands that sprung up after the mine water began to discharge in 1979 was not required. In theory the clean polishing pond at the end of the treatment system could replace (in terms of mitigation credits) the contaminated volunteer wetlands that were destroyed during construction of the treatment system.

B.2.7 Stakeholder Challenges

A century of hard rock mining in the Tri-State Mining District resulted in substantial environmental degradation of land and water resources. Although the region was designated as a suite of USEPA Superfund sites in the early 1980s, streams draining the mining-impacted watersheds still fail to meet designated beneficial uses, in part due to administrative decisions rendering them "irreversibly damaged". However, three decades of remediation work, coupled with changes in mine pool water quality and advances in passive treatment technologies, merit reconsideration of these decisions. Furthermore, given the recent relocation of residents of the most severely impacted communities, unique opportunities exist for large-scale ecological engineering implementation projects that could effectively address water quality. Initial results from a pilot passive treatment system are promising and fit well with possible future land uses being considered by the Quapaw Tribe of Oklahoma (i.e., a return to native prairie and associated wetland and riparian habitats). Successfully addressing water quality higher in the watersheds would also provide substantial downstream benefits to other Native American tribes, residents and fish and wildlife resources.

An 'attractive nuisance' for wildlife, is an issue resulting from the creation of the treatment system and potential exposure of wildlife to contaminants in the water and sediment of the unit cells. The most likely exposure would occur in the water of the aerobic wetland and the BCR but this is thought to be much less contaminated with metals than the previous volunteer wetlands.

B.2.8 Other Challenges and Lessons Learned

There have been several instances where the mine water backed up in the treatment system and the emergency bypass had to be activated to divert some of the water to the un-named tributary of Tar Creek to avoid overflowing unit the berms while work was conducted to remedy the problem. In one instance the system effluent pipe had floated out of the water, shutting off the outflow and caused water to back up in the system. This circumstance was identified during one of the routine inspections. It was promptly repaired (with coordination and assistance from the City of Commerce) and the water was re-directed into the treatment system as very short time after discovery.

Another instance when the water backed up in the system to the point of nearly overflowing the berms occurred during the first spring of operation. Some of the water was diverted to the emergency bypass to evaluate the cause of the high water levels in ponds 1 and 2. The piping was investigated as a potential cause of the problem. Also the Agra-drain panels were adjusted in an attempt to increase the water flow out of the ponds. This is a longer term iterative process in which the response of the change takes many days to equilibrate to the new flow conditions. The pipes were cleaned out. A muskrat was found partially clogging one of the pipes and accumulation of subaqueous vegetation in the water conveyance structures may have added to the reduced outflow of water. The piping system was modified by adding animal guards (e.g., varmint grates) to the pipe ends and straightening the piping diagram by eliminating a number of elbows and 90 degree joints in the 6 inch pipe to reduce the head loss they contributed. The system was placed back in full service. As a result, inspection and cleaning of vegetation from the inflow and outflow structures (i.e., varmint grates and Agra-drains) is included in the routine monitoring and maintenance.

It was found that the aerators work better if suspended in the water column rather than resting on the bottom of the re-aeration ponds. They were suspended using children's water noodles of various colors. Apparently turtles like the green ones as they have eaten them to the point of sinking the aerators. Muskrats are also a problem with burrowing in the berms and clogging the piping.

The YSI multi-meters installed at continuous monitoring sites are inspected during periodic monitoring and are serviced annually. An optical dissolved oxygen probe with a wiper is now used on the YSI multi-meter since the membrane probes tended to foul often from iron oxyhydroxide precipitates.

Additional artesian discharges are found in the upper Tar Creek and Beaver Creek watersheds (Figure B.2-1). However, contaminant concentrations and mass loadings are less than or similar to these untreated discharges (Table B.2-4), making passive treatment an especially attractive option to address surface water quality degradation.

Table B.2-4. Comparison of water quality characteristics for existing passive treatment system design and other potential discharges to be treated (adapted from Nairn et al. 2011)

Parameter	Existing passive system design influent	Representative Beaver Creek artesian discharge	Estimated Tar Creek artesian Discharge at Douthat
pH	5.95	6.62	6.2
Total alkalinity (mg/L)	393	185	100
Net alkalinity (mg/L)	29	90	
Fe (mg/L)	192	9.28	55.1
Zn (mg/L)	11	1.60	8.05
Ni (mg/L)	0.97	0.02	0.235
Cd (µg/L)	17	1	0.004
Pb (µg/L)	60	15	0.003
As (µg/L)	64	20	0.014
SO_4^{-2} (mg/L)	2239	244	1231
Discharge rate (LPM)	1000	25	8500

The size of the BCR unit may be reduced compared to this pilot system (which was designed using the 0.3 mole / m^3 – d sulfate reduction rate applied to the mine water discharge outflow), if the design were based on the metals loadings after the oxidation pretreatment step. Cadmium and lead and to a lesser extent zinc and nickel loads are reduced due to co-precipitation in the oxidation pond and aerobic wetland. This along with better post treatment re-aeration may reduce the initial high sulfide odors.

B.2.9 References

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B.3 Haile Mine Case Study

The mining team would like to acknowledge Paul Eger, Formerly with Minnesota Pollution Control Agency and James J. Gusek, Sovereign Consulting who submitted the Biochemical Reactors Case Study.

B.3.1 Site Information

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Name, Location, and Site Description

The Haile Mine site is located in Lancaster County, South Carolina, approximately three miles north of Kershaw in the north central part of the state (Figure B.3-1). Gold has been mined in the region since the early 1800s. With its history dating back to 1827, the Haile Mine is the second oldest significant gold mine in the southeastern United States. Historical mining began with the recovery of placer gold from the gravel deposits of Haile Gold Mine Creek and advanced into small open pits during the 1830s and 1840s. Haile produced sulfur for explosives and medicines during the civil war until portions of the mine buildings were burned to the ground in 1865 by Union forces under General Sherman's command. Intermittent gold mining continued from 1870 through 1942, at which time open pits were expanded and localized underground workings were developed (Golder 2004).

Modern open pit mining and heap leach gold recovery resumed in 1984 and ceased in 1992. Since then, various site reclamation activities have successfully reclaimed both modern facilities and historical mining features that predate the first modern-day production in 1985. These reclamation/closure activities include backfilling depleted open pits and capping rock dumps.

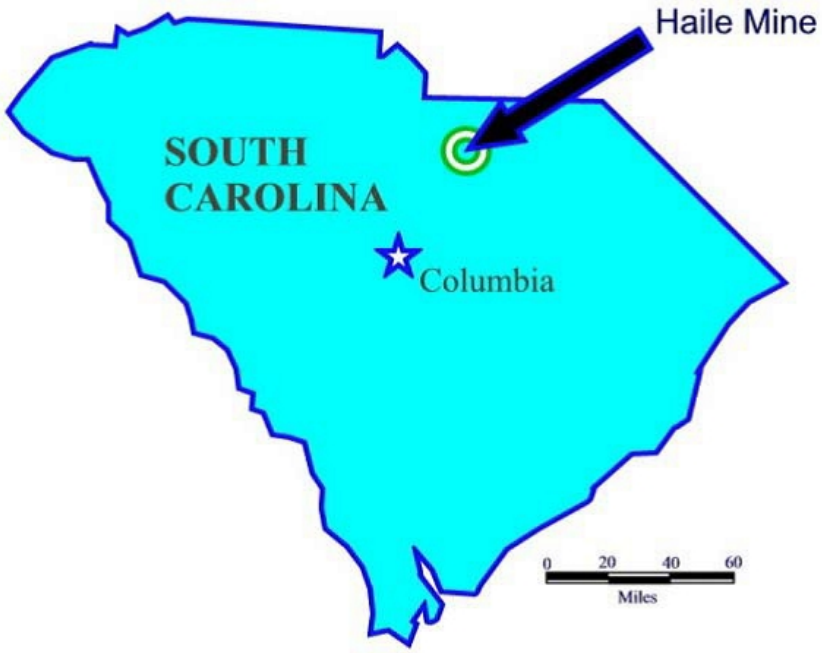


Figure B.3-1. Location of Haile Mine, South Carolina, USA

Following the latest episode of site closure work, mining-influenced water (MIW) requiring treatment originated from three sources (see Figure B.3-2):

- A backfilled and capped Chase Hill open pit with drainage of about 19 liters per minute (L/min) [5 gpm],
- The closed South Heap Leach Pad (South Pad), which was capped in 2000, with an MIW flow ranging from 0.15 L/m (0.04 gpm) to about 1.9 L/m (0.5 gpm), and
- The closed Chase Hill Pad was re-graded and capped with compacted clay in 1999 which was upgraded to a geomembrane system in 2005. A peak MIW flow of 0.76 L/min (0.2 gpm) or less and periods of near zero flow associated with dry periods is expected.

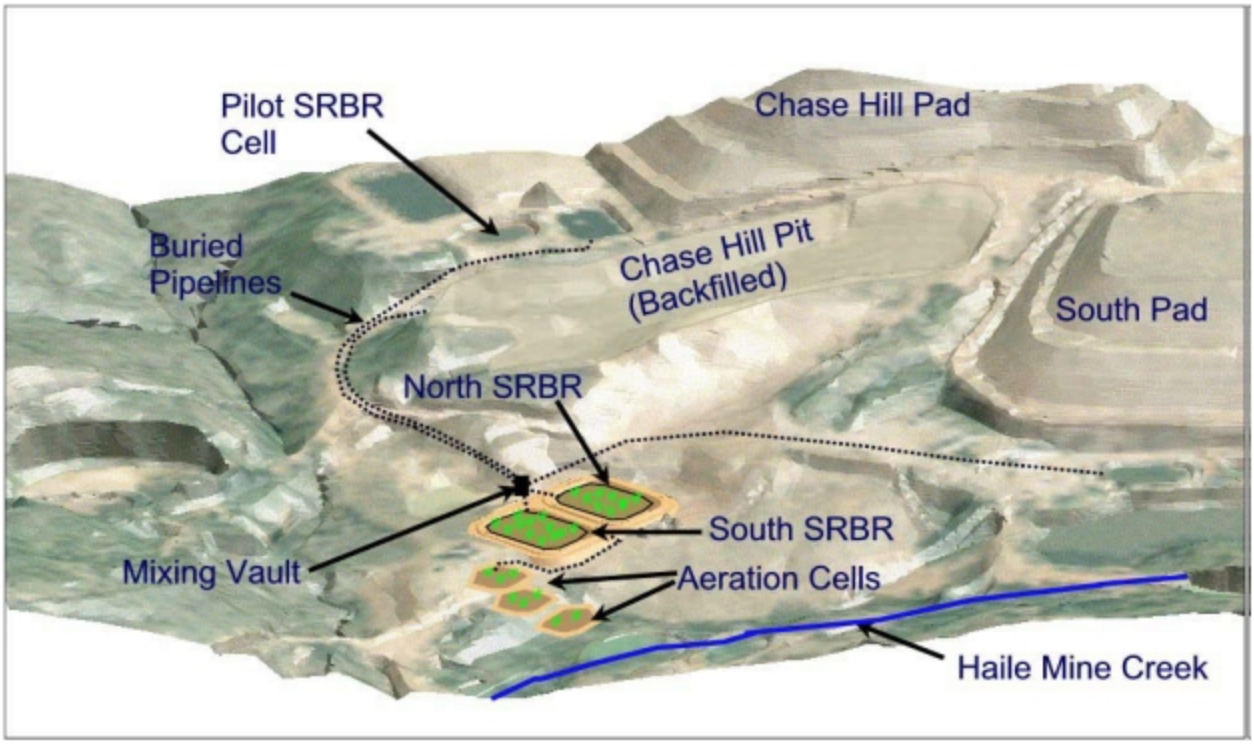


Figure B.3-2. Proximity of reclaimed mining areas in relation to as-built SRBR system lay-out.

B.3.2 Mining-influenced Water Chemistry

The water quality of flows from the Chase Hill Pit, Chase Hill Pad, and South Pad is shown in Table 1; flow rates vary among the water sources. Table 1 also presents the projected values for selected water quality parameters after mixing the three sources into a combined water sample. These estimated values were based on a weighted-average calculation. The individual weighting was proportional to the projected flow rate from the given MIW source. The actual combined concentrations of the various parameters in Table B.3-1 reflect the beneficial effects of implementing acidic MIW prevention measures such as capping of net acidic mine wastes and backfilling pits. Updated data from individual MIW sources is not available but it is suspected that the mass contribution from the Chase Hill Pad was significantly reduced when a geomembrane cap was installed over it, coincidentally completed when the passive treatment system was commissioned.

Table B.3-1. Individual and combined chemistry of closed Haile Site MIW– projected & actual

Parameter	Closed South Pad	Closed Chase Hill Pit	Closed Chase Hill Pad	Estimated Combined Flow into SRBR System	Actual Combined Influent Chemistry (4.5 yr average)
Flow, L/min	1.9	18.9	0.76	22	22.1
Flow, gpm	0.5	5.0	0.2	5.7	5.6
pH	4.9	3.65	2.0	3.3	3.4
Fe, mg/L	1,290	60	11,000	552	96
Cu, mg/L	2.00	0.3	18	1.1	0.22
Zn, mg/L	2.4	0.5	37	1.9	1.8
As, mg/L	0.0	0.0	100	3.5	0.09
Ni, mg/L	2.3	0.16	16.7	0.9	No data
Co, mg/L	0.0	0.0	13.30	0.5	No data
Al, mg/L	539	17	1,700	122	24.7
Mn, mg/L	-	-	-	5.9	2.8
Sulfate mg/L	30,000	750	30,000	4,342	554
Acidity mg/L	6,830	195	35,000	1,998	387
Alkalinity mg/L	0	0	0	0	0

B.3.3 System Design

Design of the passive SRBR system was based on the performance of a 2.5 year pilot-scale test and was further supported by two supplemental bench-scale studies. The design was developed in accordance with established engineering protocols; the construction was authorized under the Construction Permit 18,873-IW issued on June 15, 2004 by the South Carolina Department of Environmental Control.

A schematic of the system is provided on Figure B.3-3. Note that the vegetation drawn on the surfaces of the two SRBRs is slightly different. The South SRBR plant community is dominated by willows (Salix); the North SRBR plant community appears to be dominated by cattails (Typha). These supposedly minor differences may account for significant variations in the relative performance of the two SRBRs. The colonization by neither of these two plant communities was intentional; their establishment occurred on a volunteer basis.

The pilot sulfate reducing bioreactor (SRBR) cell test results (Golder 2004) demonstrated that the system has resiliency during extended periods of overloading. In the 4.5 years of actual operation, the system does not appear to have been exposed to overloading conditions. If anything, the system appears to be under-loaded with respect to its design conditions as suggested in Table B.3-2 and subsequent performance data.

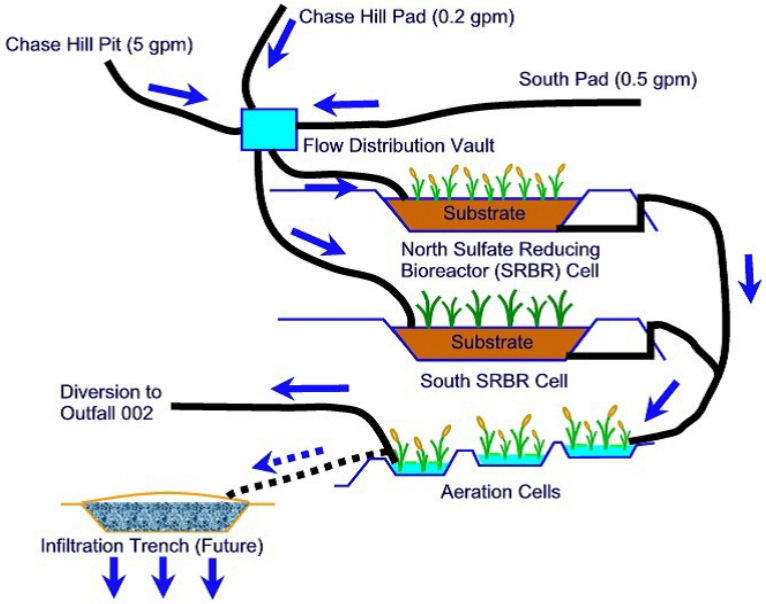


Figure B.3-3. System schematic layout

B.3.4 BCR Design

The design objectives of the SRBR cells included accommodating and managing about 22.7 L/min (6 gpm) of combined MIW flow, operating continually without pumps and allowing periodic (multiple decade) major maintenance operations. The SRBR cell design criteria were established in bench- and pilot-scale tests. These include satisfying a volumetric metal loading factor of 0.3 moles of metal loading per day per cubic meter of organic substrate (Wildeman et al. 1993), and a bottom area hydraulic loading factor of about 87.2 m^2 per L/min (3,550 square feet per gpm) of flow based on the findings of bench and pilot studies (Golder 2004). Installing just 0.915 m (3 ft) of organic substrate would have satisfied the metal loading design criteria. To be conservative and to prolong the interval between major SRBR retrofitting/substrate replacement, the actual installed organic substrate thickness was 1.68 m (5.5 ft). Thus, if the design water chemistry and design flow rates were maintained, the volumetric loading "as-built" would be about 0.16 moles of metal/day/m^3. Design issues typically included in SRBR based systems are discussed in Gusek (2002).

The ratio of components in the organic substrate mixture as cited in Golder 2004 was changed just prior to construction in response to localized procurement shortages. The final substrate mixture is provided in Table B.3-2.

Table B.3-2. As-installed organic substrate

Mixture	Cubic Meters		Mixing Ratio
Material			
Aged Chipped Wood	184	or	20 buckets at 9.2 m^3/bucket
Manure	9	or	1 bucket at 9.2 m^3/bucket

Mixture	Cubic Meters		Mixing Ratio
Agricultural Lime-stone	14	or	6 buckets at 2.3 m^3/bucket
Hay / Alfalfa (bale)	21 bales	or	At 309 kg (680 lbs) per bale

To minimize earthwork, provide for future maintenance, and conform to the existing surface contours at the proposed site, the SRBR treatment capacity required was allocated equally between two individual cells which were subsequently operated in parallel. Each cell was designed to accommodate a nominal flow of 11.4 L/min (3 gpm); however, as demonstrated in the pilot test program, each cell could handle as much as three times that MIW flow for an extended period with a small decrease in removal efficiency.

Flow through the SRBR cells was arranged to be by gravity from top to bottom. The cells were lined with 60 mil thick linear low density polyethylene (LLDPE) geomembrane. Under typical operating conditions, the organic substrate would be completely submerged beneath a standing pool of water, similar to the operation of the pilot cell.

The water exiting from both SRBR cells is commingled in a buried pipe and conveyed to the uppermost portion of an aerobic polishing cell (APC). The APC receives treated effluent from the two SRBR cells. The primary purpose of these units is to:

- provide re-oxygenation
- remove any remaining dissolved ferrous iron
- remove a minor loading stream of dissolved manganese (about 2.8 mg/L)

The APC delivery pipe terminus was equipped with an upturned 90-degree elbow to preclude small animal incursions. The peak flow, ranging from 12.1 to 28.4 L/min. (3.2 to 7.5 gpm), is so small that beavers known to live in nearby natural drainages have not been attracted to the location.

Construction began in late October, 2004, and was substantially complete by April, 2005; monitoring of the system commenced in May, 2005. Photos of the North and South SRBR cells are provided in Figure B.3-4 and B.3-5, respectively.



Figure B.3-4. North SRBR (April 2008) showing cattail (Typha sp.) as major vegetation.



Figure B.3-5. South SRBR (April 2008), showing willow (Salix sp.) as major vegetation.

B.3.5 BCR Performance

Since 2004, the Haile Mine passive SRBR system has functioned 24 hours a day, seven days a week without sustained interruptions. SRBR and APC performance data are shown for Figures 6 through 13. Discussions of individual parameters of interest follow.

Previously, it was noted that the SRBRs were designed for about 0.16 moles of metal removal per day per cubic meter of substrate, significantly less than the benchmark value of 0.3 moles/day/m^3. The data plotted on Figure B.3-6 show that both SRBRs were not excessively loaded with respect to metals during the five years of operation and that sulfate removal more or less kept pace with metals removal.

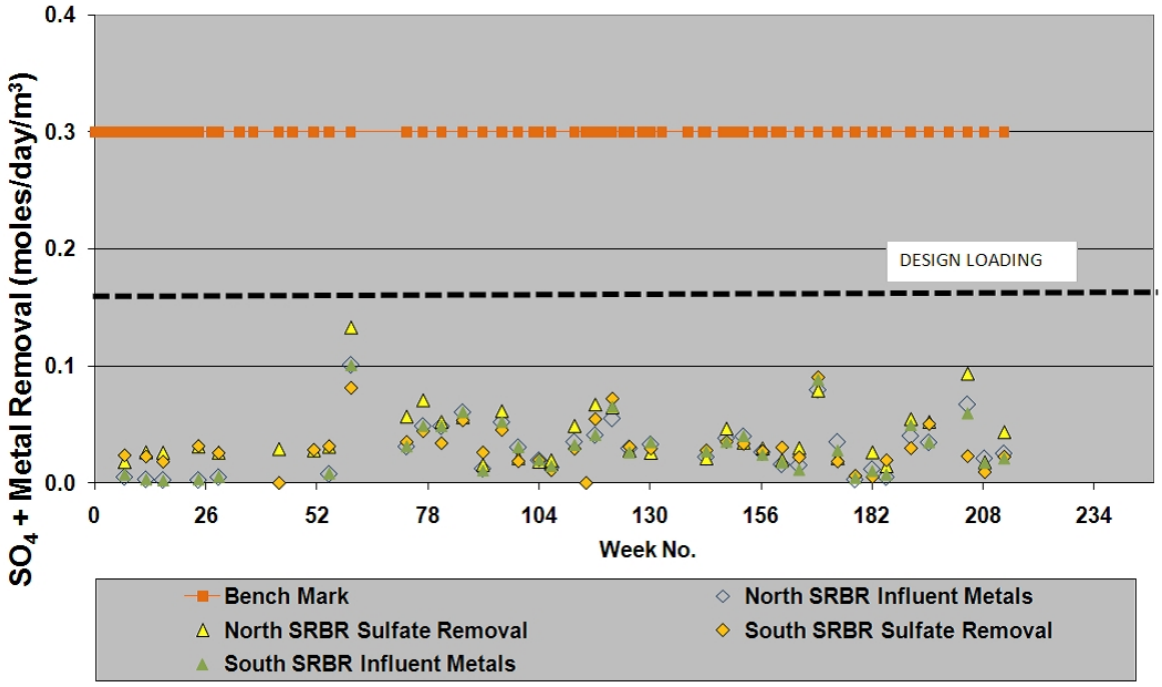


Figure B.3-6. Volumetric sulfate and metal loading/removal.

B.3.5.1 pH Improvement

The data in Figure B.3-7 suggest that the pH improvements were relatively insensitive to seasonal changes for the five years the SRBR system has been in operation. The rise in Aeration Cell pH compared to either of the SRBR cell effluents would suggest that excess alkalinity in the SRBR effluent (typically about 800 mg/L for the SRBR) provided a consistent buffering effect. In comparison, the average net alkalinity in the Aeration Cell was about 583 mg/L.

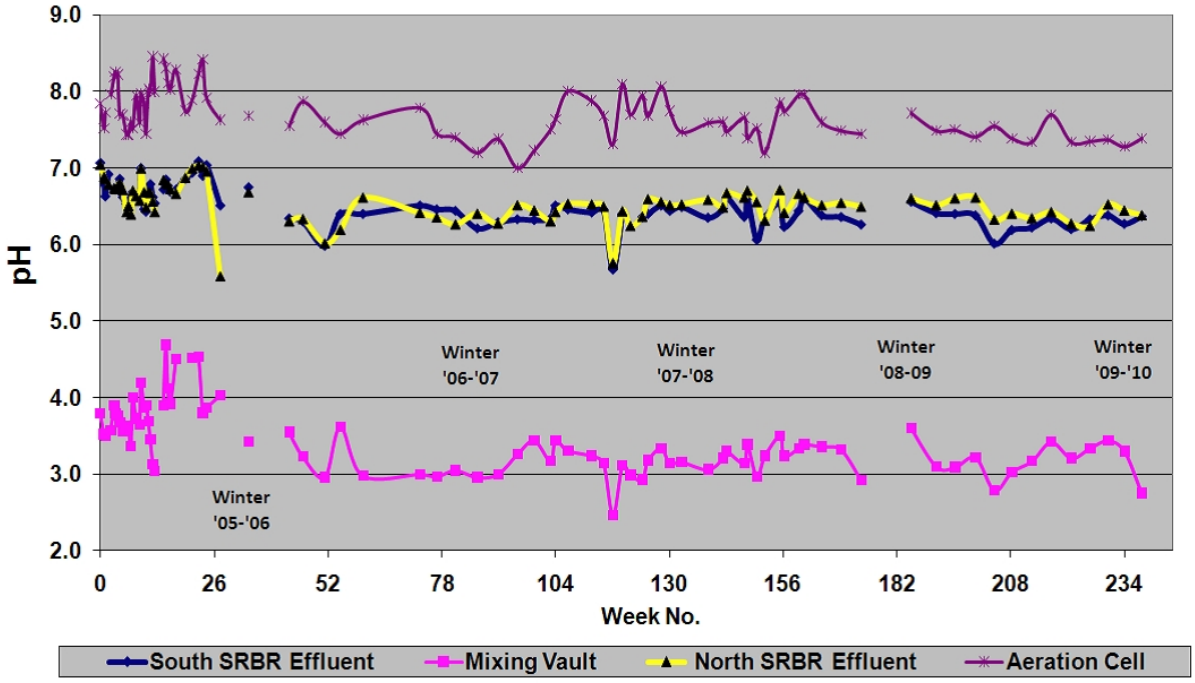


Figure B.3-7. pH.

B.3.5.2 Oxidation Reduction Potential (ORP)

Once the noisy ORP data from the first few months is filtered from Figure B.3-8, it is apparent that the North and South SRBR ORP data typically agree. With minor exceptions, ORP is usually negative which is indicative of robust geochemically reducing conditions. It is curious to note that the Mixing Vault ORP values decrease over time, suggesting that the mitigation measures implemented at the Haile Mine (such as capping or re-vegetation) appear to be decreasing the oxidation kinetics in the mine waste zones that the MIW contacts prior to entering the SRBR system. The typically depressed ORP in the Aeration Cell effluent suggests that the APC component of the system is being stressed. Future designs for aeration cells would probably result in larger footprints.

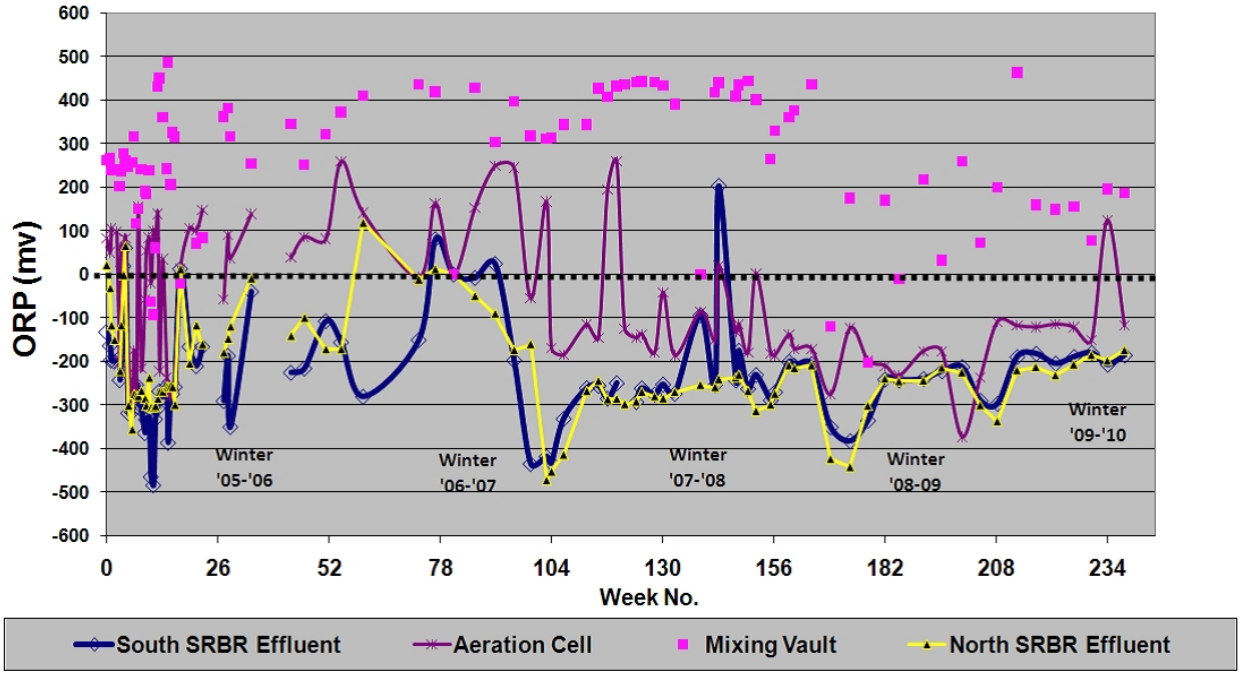


Figure B.3-8. Oxidation reduction potential (ORP).

B.3.5.3 Dissolved Iron

As plotted in Figure B.3-9, dissolved iron removal in the North and South SRBRs was virtually identical for weeks 0 through 84, which coincides with the spring of 2007. At this point, the behavior of the two SRBRs diverges, and the dissolved iron removal performance of the South 427 SRBR appears to become worse over time. It is curious that the North SRBR appears to have had a similar excursion in about week 182, just prior to the onset of the 2008-09 winter.

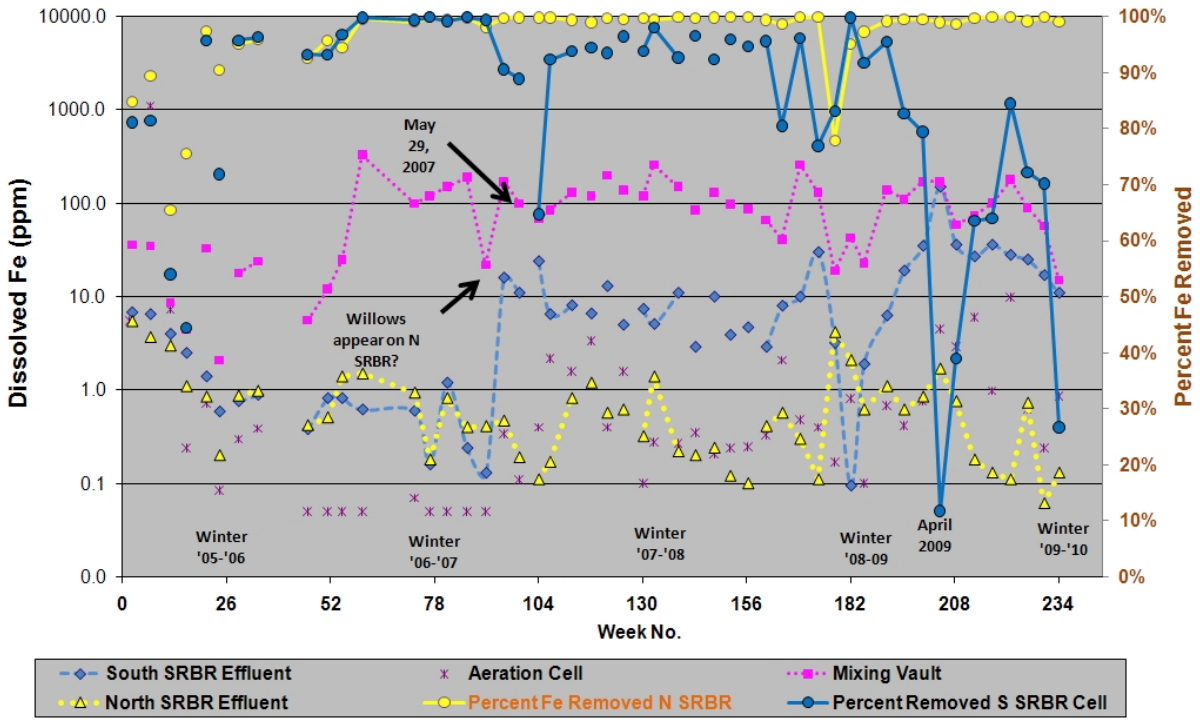


Figure B.3-9. Dissolved iron.

Based on recollections from site personnel, it appears that willows first appeared on the surface of the South SRBR in the spring of 2007, which coincides with the drop in the South SRBR iron removal performance. Similar excursions appear to occur with a somewhat inconsistent regularity when the seasons change. The current hypothesis is that the ORP conditions on the surface of the South SRBR are changing in response to the dormancy or spring emergence of the willow community with the changing seasons. For example, in the autumn, when dormancy begins, oxidizing conditions typically surrounding the plant roots may become reducing. Any iron oxy-hydroxide that had been precipitated in this zone would likely be re-dissolved in the reducing conditions, creating a temporary spike in iron concentration in the SRBR effluent as Fe2+ iron. During the winter, reducing conditions would continue to prevail and ferrous iron would likely be removed in response to bacterial sulfate reduction as iron sulfide. In the spring, when the willows are no longer dormant, the iron sulfide precipitates would now be exposed to oxidizing conditions in the willow root zone and again, iron would be released when the sulfides were oxidized. This phenomenon was first observed in the iron removal data for the lime dosed aerobic cell in the Wheal Jane passive treatment system as reported by Hamilton et al. (1997). Similar spikes were observed in December of 1995 and the following spring of 1996. This aerobic cell was planted with three species of plants: Typha, Phragmites, and Scirpus.

One might expect to see a coincidental spike in arsenic in the South SRBR effluent but none was detected. Mixing Vault arsenic levels ranged from 0.007 to 0.39 mg/L and both South and North

SRBR effluents consistently exhibited arsenic concentrations less than the detection limit of 0.005 mg/L.

B.3.5.4 Dissolved Copper

As plotted in Figure B.3-10, dissolved copper removal in the North and South SRBRs (plotted cumulatively) appears to coincide with the iron removal anomalies identified in Figure B.3-9. However, these "false" anomalies are attributable to depressed copper levels in the Mixing Vault and the nearly constant analytical detection limit for Cu of 0.01 mg/L. For example, the Mixing Vault copper concentration on Oct 28, 2008 was only 0.029 mg/L. With a detection limit of 0.01 mg/L, this results in an apparent removal of only 52.6% [(0.029-0.01)/0.029 = 0.526]. 429.

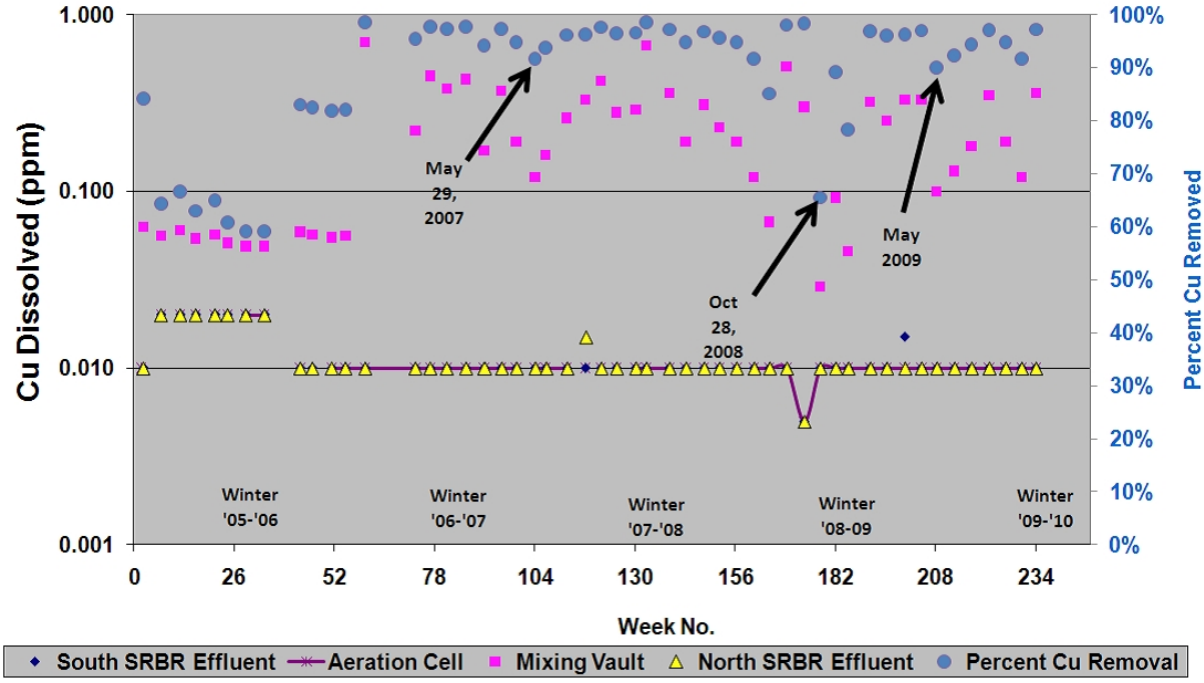


Figure B.3-10. Dissolved copper removal.

B.3.5.5 Dissolved Zinc

A similar masking mathematical influence was observed in a plot of Zn removal, as shown in Figure B.3-11. Consequently, it is concluded that the willows or some other unidentified mechanism is affecting iron removal efficiency; it is not significantly affecting Cu or Zn removal. Neither dissolved Cd removal efficiency (typically about 94%) nor dissolved Al removal efficiency (typically about 97.5%) appeared to vary seasonally in the South SRBR.

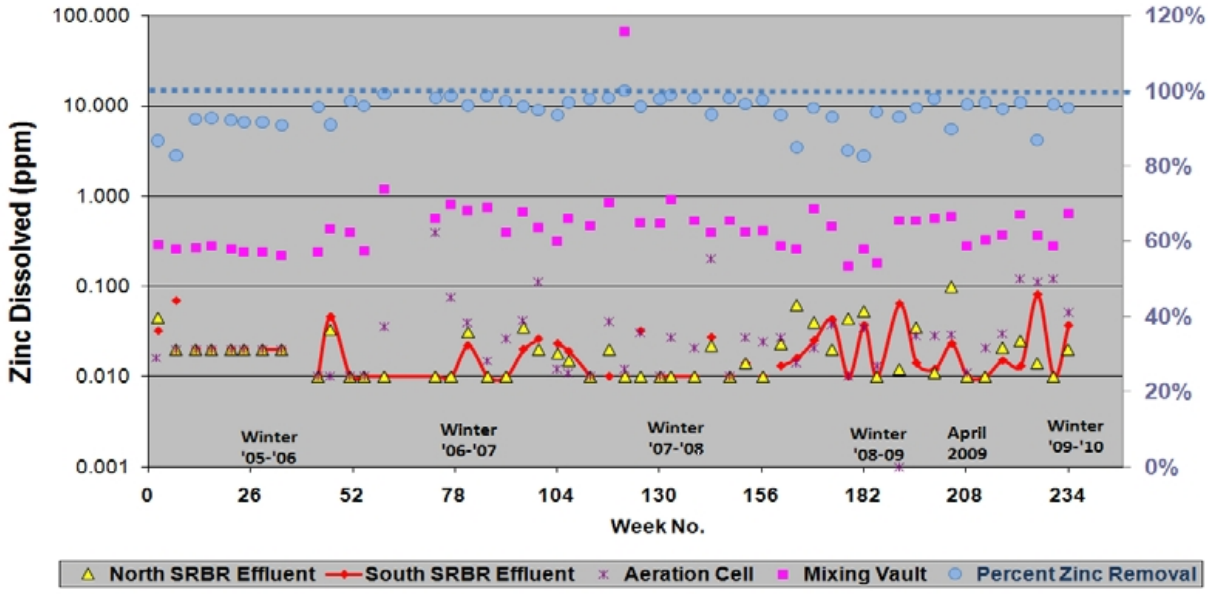


Figure B.3-11. Dissolved zinc removal.

B.3.5.6 Sulfate Reduction

Sulfate removal is naturally a key indicator of the relative "health" of a sulfate reducing bioreactor. Data in Figure B.3-12 reflect a wide scattering of influent sulfate concentrations in the Mixing Vault and relatively consistent effluent concentrations of typically less than 600 mg/L. It is curious to note that the relative difference in sulfate concentration between the influent and the SRBR effluent is about 300 mg/L in the first year of operation. Subsequent yearly peak data suggest differences up to about 900 mg/L. However, when this data is synthesized with volumetric loading, as plotted in Figure B.3-6, the data trends are far less noisy.

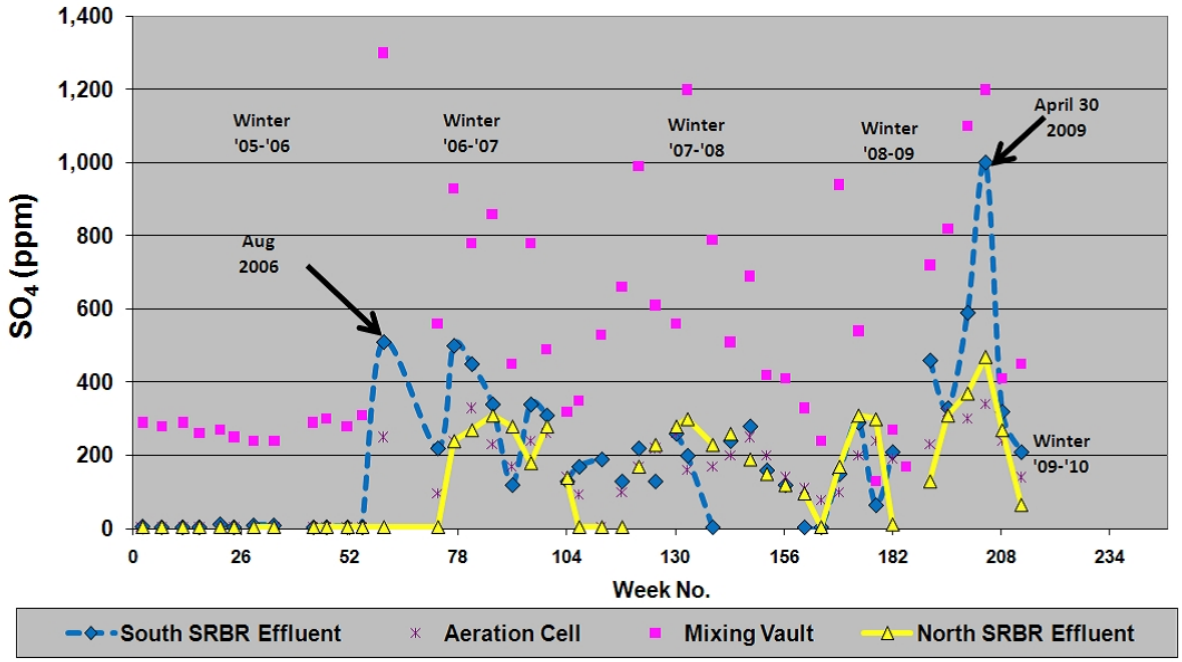


Figure B.3-12. Sulfate removal.

B.3.5.7 Combined Metals Removal Efficiency

A final measure of success of a passive treatment system is the percentage removal of the metals of interest. Figure B.3-13 reflects the relative removal efficiencies of the two SRBRs and the final aeration cell in removing iron, aluminum, copper, zinc, and cadmium. The alkalinity buffering in the SRBR effluent is likely countering the adverse iron performance in the South SRBR to provide its intended polishing effect.

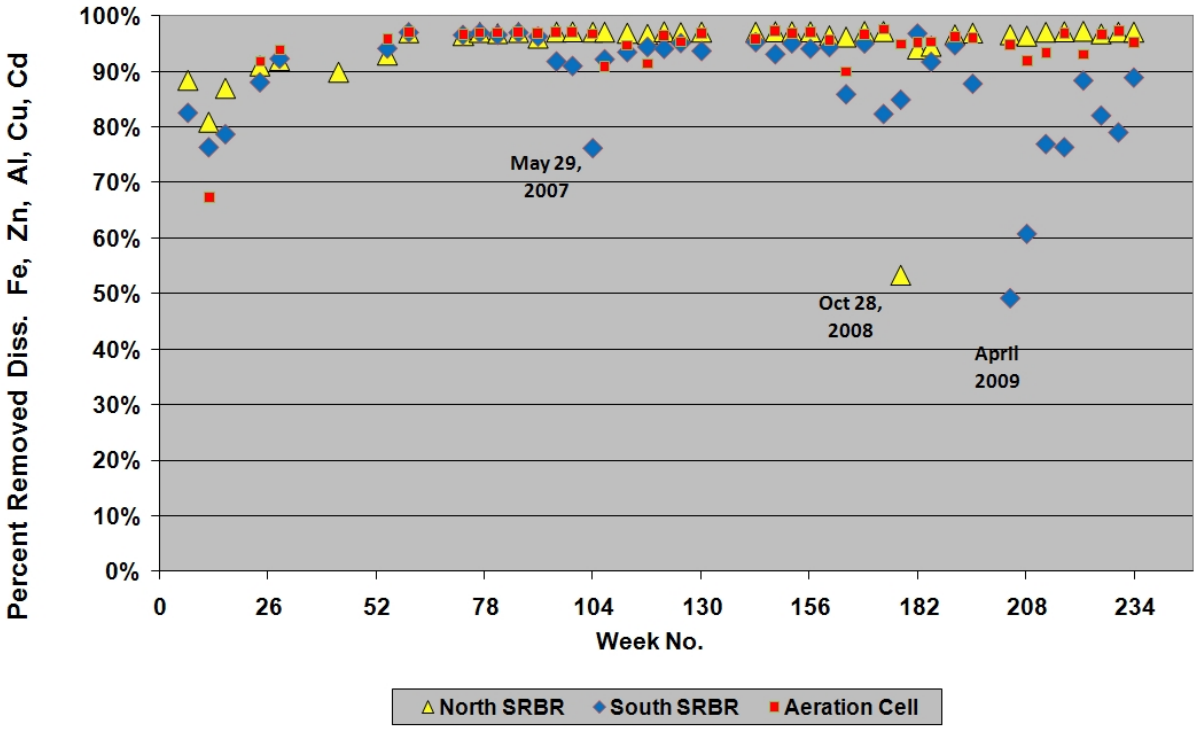


Figure B.3-13. Combined dissolved metals removal efficiency, SRBRs and aeration cell.

B.3.6 Monitoring

Influent and effluent samples were collected about once per month from 2005 to early 2010. All samples were analyzed for pH, Eh, conductivity, SO_4, copper, zinc, nickel, cadmium, arsenic, iron, manganese, aluminum and conductivity.

B.3.7 Challenges and Lessons Learned

No regulatory or stakeholder challenges were reported. Decreased removal of iron in the south SRBR corresponded with the appearance of willows on the top of the cell. Typically vegetation is not encouraged due to concerns with oxygenating the water and decreasing the reducing zone. Vegetation on the north SRBR was dominated by cattails and dissolved iron removal was not affected.

Substrate samples collected from the preceding pilot test showed that concentrations for the parameters of interest were elevated only in the upper portion of the substrate (Table B.3-4)

Table B.3-4. Biopsy samples of substrate from pilot cell

	Sample Depth (inches)	Total Metals Concentration (mg/kg)	% Organic Matter	% Solids		Sample Depth (inches)	Total Metals Concentration (mg/kg)	% Organic Matter	% Solids		Sample Depth (inches)	Total Metals Concentration (mg/kg)	% Organic Matter
	From	To	Al	As	Cd	Ca	Cu	Fe	Mg	Mn	Zn		
Sample	0	6	23,300	1.08	6.37	17,300	253	25,900	8,020	264	844	64.3	43.4
Zone 1	6	12	7,680	0.89	4.39	69,200	89	20,200	31,600	484	915	47.1	42.6
	12	18	5,280	0.92	2.23	56,000	48	13,100	22,900	474	458	65	43.8
	18	21	2,500	0.79	1.72	58,300	30	6,830	23,800	469	434	65.8	42.9
Sample	0	6	19,700	0.93	6.54	27,500	285	36,400	13,600	245	901	62.8	41.8
Zone 2	6	13	14,400	1.16	5.42	33,800	224	32,500	16,300	340	789	59.3	39.8
	13	18	3,310	0.51	1.42	41,500	26	13,700	17,200	359	271	72.7	43.3
	18	24	2,800	0.56	1.38	51,700	27	10,100	20,800	351	298	63.8	44.4
Sample A[1]			14,900	1.38	6.83	25,500	380	40,100	10,700	351	969	60.8	42.3
Baseline[2]			9,730	5.98	3.33	60,500	53	7,380	27,300	505	958	78.7	84
Ag Lime						185,000			106,000				

Notes:
[1] Sample A was a composite of the outer 1/2" of substrate in direct contact with the cell sidewall.
[2] Unexposed substrate

B.3.8 References

Golder Associates Inc. 2004. Haile Mining Company Inc. Passive Best Management Practice System Design Report, dated February 2004.

Gusek, J.J., 2002. Sulfate-Reducing Bioreactor Design and Operating Issues: Is This the Passive Treatment Technology for Your Mine Drainage? Presented at the National Association of Abandoned Mine Land Programs (Park City, Utah, September 15-18, 2002).

Hamilton, Q.U.I., H.M. Lamb., C. Hallett, and J.A. Proctor, 1997. Passive Treatment Systems for the Remediation of Acid Mine Drainage at Wheal Jane, Cornwall, UK. In: Proceedings of a CIWEM National Conference, (Newcastle University, September 1997), pp. 33-56.

Wildeman, T.R., G.A. Brodie, and J.J. Gusek. 1993. Wetland Design for Mining Operations. Vancouver, B.C.: BiTech Publishing Co.

B.4 Ferris-Haggerty Mine Case Study

The mining team would like to acknowledge James J. Gusek, Sovereigh Consulting who submitted this Biochemical Reactors Case Study.

B.4.1 Site Information

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Name, Location, and Site Description

The Ferris-Haggerty mine is a 100 year old abandoned underground copper mine on private land within the boundary of the Medicine Bow National Forrest near Encampment, Wyoming at an elevation of 9,500 feet. The site is situated in the remote upper reaches of Haggerty Creek, about four miles north of Wyoming Highway 70 in Section 16, T14N, R86W (Figures B.4-1 and B.4-2). At the turn of the nineteenth century it was one of the largest Copper mines in the region, producing an estimated 24 million pounds of Copper from 1898 to 1908. The mine used state of the art mining equipment, including a 16-mile aerial tramway (touted as the largest in the world at the time) for transporting metal ore to a smelter in Encampment.

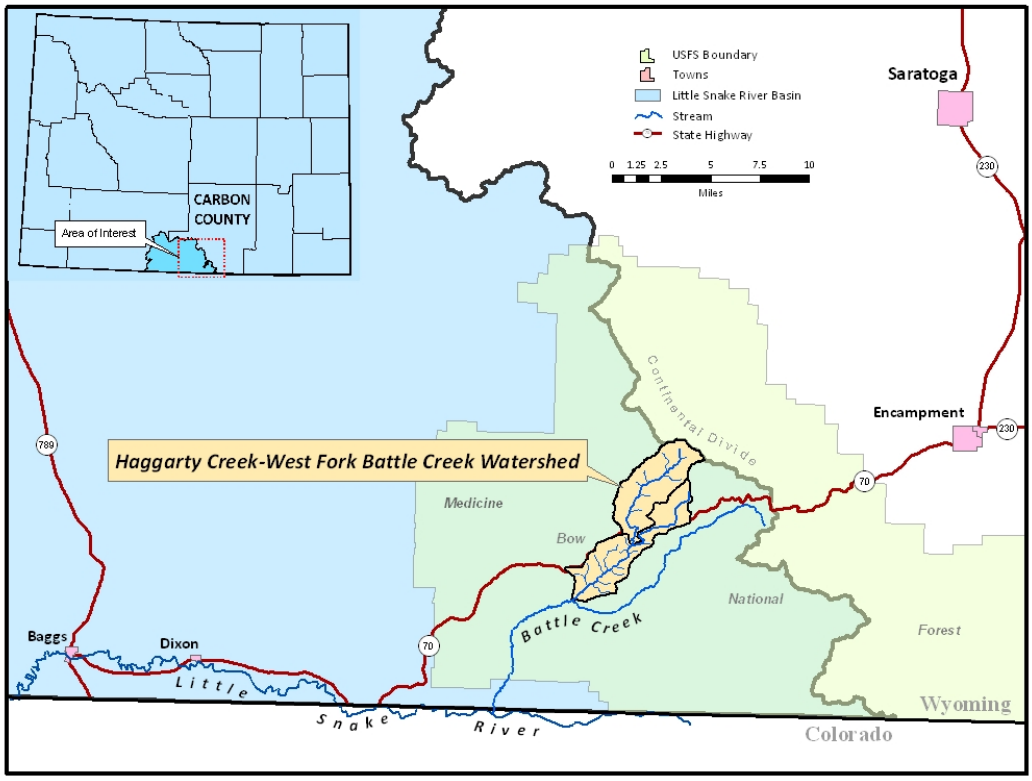


Figure B.4-1. Haggerty Creek-West Fork Battle Creek Watershed, Carbon County, Wyoming.

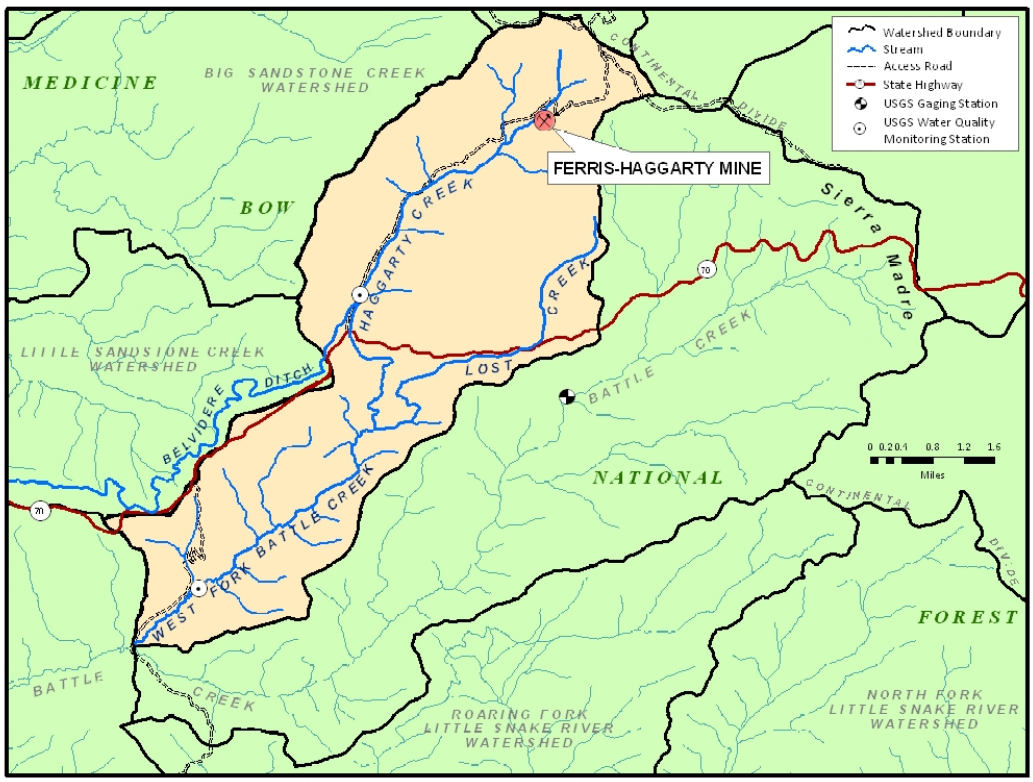


Figure B.4-2. Haggerty Creek-West Fork Battle Creek Drainage, Carbon County, Wyoming.

The copper ore was initially extracted from the underground workings through two vertical shafts driven from the surface and later from a main haulage level, called the "Osceola Tunnel", whereby the ore was transported to the surface using a compressed air locomotive and rail cars. The ore was then transported over the Continental Divide to the smelter in Encampment via the aerial tramway. By 1904 the mining operations were at their apex, employing 200 men and producing over $1,400,000 worth of copper. Mining operations ceased in 1908 presumably as a result of smelter fire, financial problems, mismanagement and declining copper prices. Foreclosure proceedings began in 1913 and salvage operations followed shortly. The Ferris-Haggerty Mine Site was added to the National Register of Historic Places on July 2, 1973.

The main environmental cleanup challenge presented by the mine is a substandard water discharge from the ore transporting/drainage tunnel. This water contains 4 mg/L of copper and discharges into Haggerty Creek at a rate ranging from 50 gallons per minute (gpm) to 500 gpm, depending upon the time of the year. This amount of copper has rendered the stream directly below the mine virtually lifeless.

The Ferris-Haggerty Mine site was investigated under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) in the early 1990s. USEPA conducted a Preliminary Assessment in 1993 and a Site Inspection in 1994 recommending further assessment

with a high priority. The State of Wyoming assumed control of the cleanup project in 1996 after the USEPA agreed with a state request not to include the mine on the USEPA's Superfund National Priority List (NPL) site. Wyoming Game and Fish Department officials have stated that cleaning up the mine site could help reestablish the Colorado cutthroat trout (endangered species) populations in nearby Haggerty Creek.

The Wyoming Abandoned Mine Land Reclamation Program (WYAMLP) spent $2.5 million at Ferris-Haggerty preparing the mine for the mitigation of acid mine drainage. The expenditure included installation of a safety closure and stabilization of the underground workings where personnel would need access. The mine's portal and main tunnel, the Osceola Tunnel, was rehabilitated and shored up to allow engineers to install a pilot water treatment systems and evaluate the high copper concentration source. A substantial locking closure was also installed at the portal and remains in place, preventing access to the mine.



Figure B.4-3. Ferris-Haggerty Mine Osceola Tunnel portal and effluent channel, September 2008.

B.4.2 MIW Chemistry

The Ferris-Haggerty Mine/Osceola Tunnel site is a high-elevation abandoned copper-mine site with acid rock drainage (ARD) characterized initially as having neutral pH, low sulfate, elevated

copper concentrations, and water temperature close to freezing. The mine water has neutral pH discharge with 3 to 20 mg/L dissolved copper and low sulfate (typically less than 100 mg/L SO_4); water temperature is close to freezing (±4˚C), and the mine is accessible only by a 32-km (20-mile) snowmobile trek for nine months out of the year. Flow from the portal varies from 57 to 114 L per min (15 to 30 gpm) in the winter months to 1.7 cu m per min (450 gpm) or more during the spring runoff. Copper concentration initially observed at the portal was fairly constant all year long, somewhat independent of flow rate. However, after the completion of tunnel rehabilitation activities, a spike of elevated copper concentration and low pH (3.8) was observed coincident with the spring freshet.

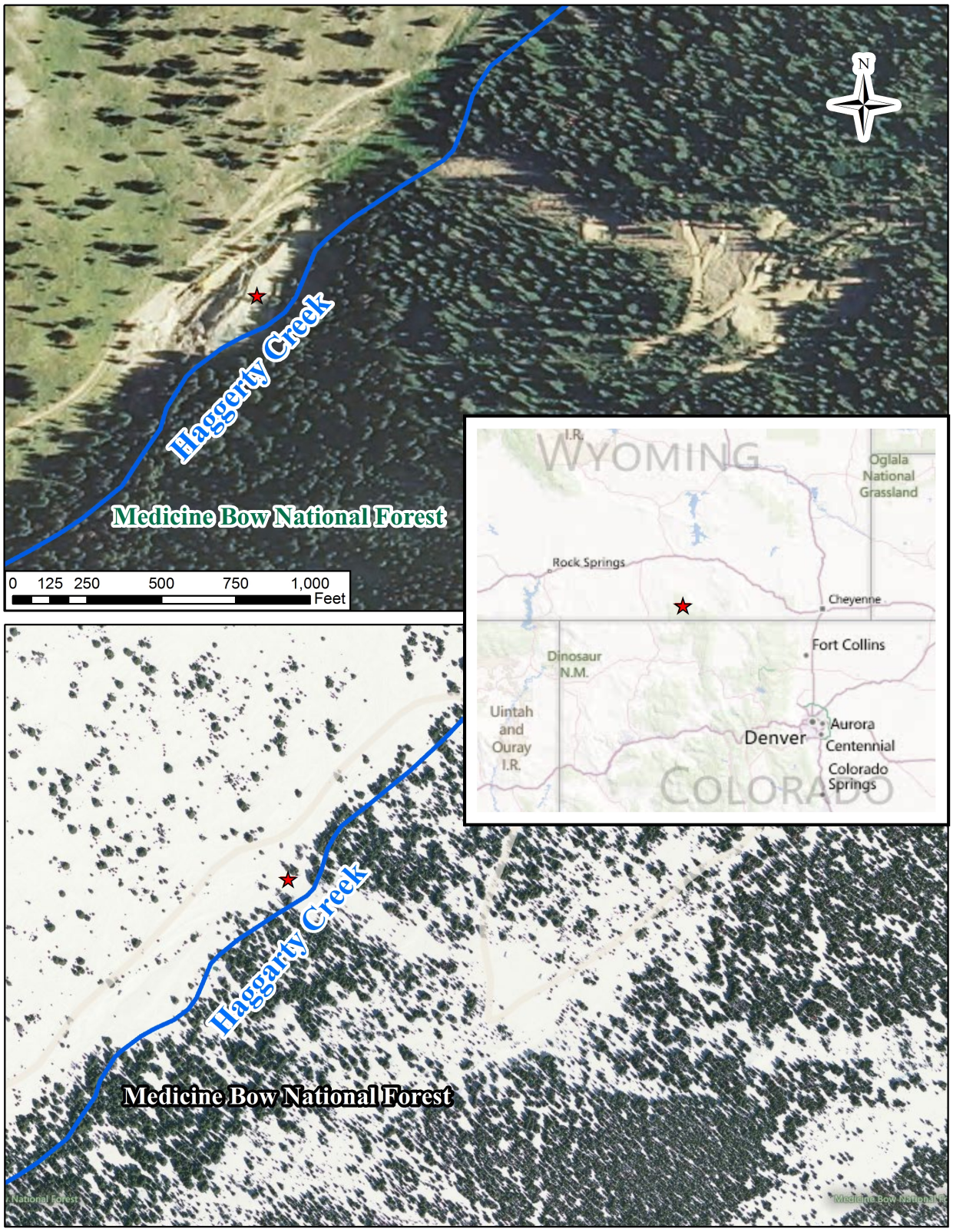

N
Haggerty Creek
Medicine Bow National Forest
0 125 250 500 750 1,000
Feet
I.R.
WYOMING
Oglala National Grassland
Rock Springs
Cheyenne
Fort Collins
Dinosaur N.M.
Uintah and Ouray I.R.
Denver
Aurora
Centennial
Colorado Springs
COLORADO
Haggarty Creek
Medicine Bow National Forest

Figure B.4-4. Ferris-Haggerty mine aerial photo location map. Top view shows site in summer months when site is accessible. Bottom view shows site during winter months.

B.4.3 System Design

B.4.3.1 BCR Design

The first of two pilot-scale passive treatment cells was composed of a single gravity-fed anaerobic SRB-cell configured in a down flow mode. In the summer of 1997, a 4.6-m (15-ft) diameter and 1.2-m (4-ft) deep cell was filled with a mixture of softwood sawdust, hay, limestone, cow manure, and gypsum. The gypsum was added as a source of sulfate for the SRB. The proportions in the recipe were selected based on a six-week-long bench-scale (trashcan-size) test conducted in the summer of 1996.

The pilot cell treated a flow of 3 to 5 gpm for two years beginning in the summer of 1997. Snow depths on the site can reach 10 feet over a typical winter season, so the cell was enclosed in a shed to allow winter access for sampling, inspection, and data retrieval.

The pilot cell was designed based on an estimated sulfate reduction rate of 0.15 moles/day/cubic meter of substrate, the value that was observed in the best cell in the 1996 bench-scale study. This results in a retention time in the cell of less than 12 hours.

The pilot cell was outfitted with an ISCO 6700 automatic sampler and a YSI probe to monitor pH, conductivity, redox potential, and water temperature in late October1997, just prior to a blizzard that effectively ended the short field season. No heat-trace equipment was installed; all pipes carrying water were buried and/or fitted to maintain flow to prevent freezing.

In August 1999, a second pilot-scale cell was built with a slightly different substrate mixture and buried below grade to more closely mimic the planned full-scale system (Gusek 2000). The full scale system was never constructed, however, due to land owner resistance. The surface area needed for construction just outside the portal was on privately owned patented mining claims and the land owner did not allow the system to be built. The WYAMLP installed a limestone line ditch from the portal to Haggerty Creek and closed the site out in mid-2000.

B.4.3.2 Pre- and posttreatment requirements

Due to the low mine water temperature, the initial slug of water used in incubation was heated to about 15˚C. The cell was incubated for about a week before it received full design flow of 19 L per min (5 gpm) at about 3˚C.

B.4.4 BCR Monitoring

Bi-monthly site visits via snowmobile were conducted during the winters of 1997/98 and 1998/99 to collect data and samples. Monthly sampling was conducted during the summers of 1998 and 1999. During the winters, sub-freezing air temperatures resulted in an ice covering over about 40

percent of the cell surface. Flow through the cell decreased from an initial value of 19 L per min (5 gpm) in October1997 to about 11.4 (3 gpm) in the subsequent months.

A number of physical and chemical parameters were evaluated for the pilot-scale cell from start-up in August 1997 through June 1999 in order to evaluate the efficiency of the pilot-scale cell. Parameters include copper removal, pH, sulfate reduction, redox potential, temperature, and saturated hydraulic conductivity.

B.4.5 BCR Performance

Figure 5 in Reisinger et al. shows total copper removal for the pilot-scale cell. Copper removal during the first nine months of operation (that is, from August 1997 through May 1998) ranged from 95 to 100 percent. Once more highly contaminated chute/shaft water (from deeper in the mine) was introduced in June 1998, copper removal efficiencies decreased to generally between 89 to 97 percent. The removal efficiencies noted during December 1998 (84 percent) and May 1999 (63 percent) were likely due to system upsets caused by contaminated water not being fed to the cell.

As shown in Figure B.4-5 below, the pH of the processed water remained relatively neutral even with a change in feed waters from relatively neutral pH (Osceola Tunnel portal water) to a pH ranging from about 3.5 to 4.0 (chute/shaft water). Both the SRB and limestone incorporated into the substrate recipe likely contributed to the ability of the cell to maintain a relatively neutral pH.

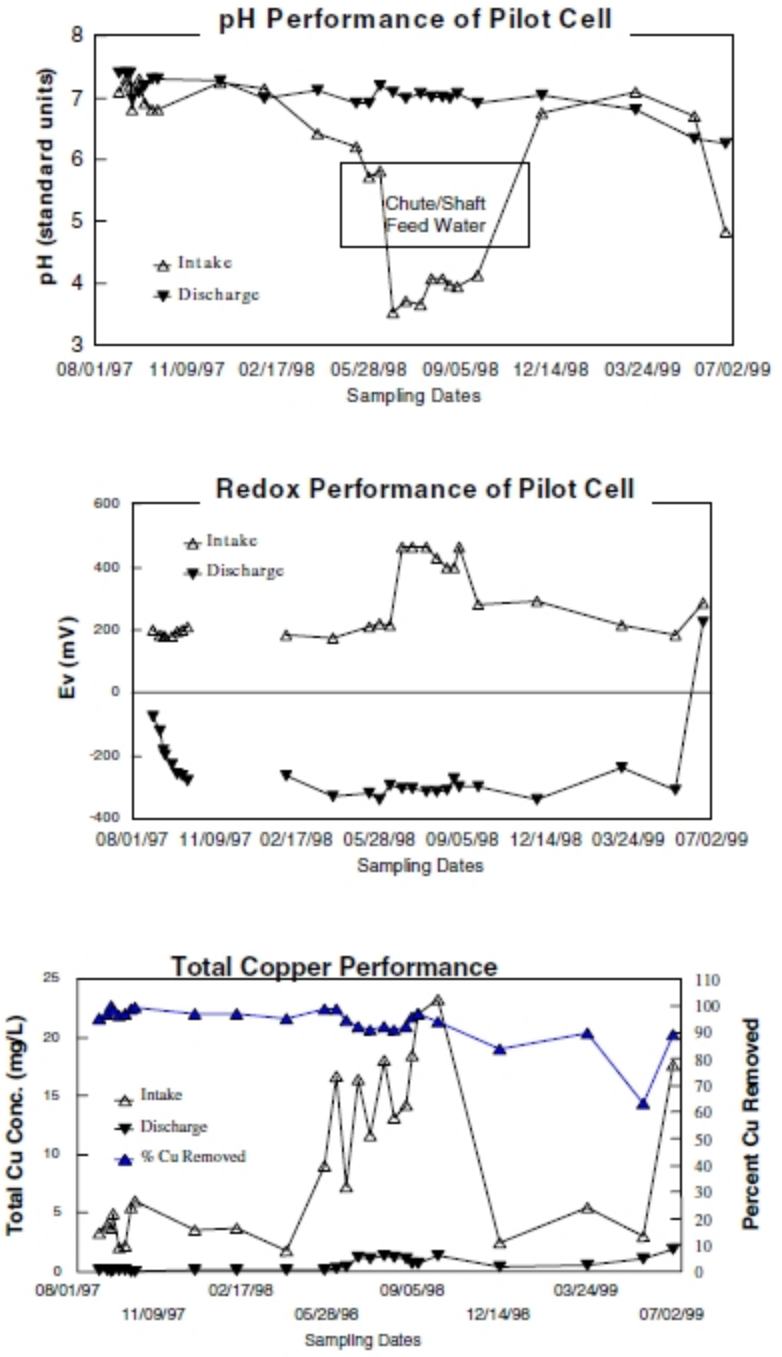


Figure B.4-5. Ferris-Haggerty mine, WY, pilot cell data.

Sulfate reduction is an indication of the level of SRB activity. The greater the sulfate reduction, the more the SRB activity. During the first several months of cell operation, sulfate reduction was not apparent and was likely masked by the addition of gypsum into the substrate recipe. Gypsum addition was done to enhance the cell environment for the SRB during system start-up. After the first

several months, the cell generally exhibited sulfate reduction, especially during the period when higher sulfate-containing chute/shaft water was being processed.

Figure 5 also shows Ev measurements for pilot cell inflow and discharge. Ev is a measure of redox potential. The more positive the Ev value, the more oxygen-enriched the water. The more negative the Ev value, the more oxygen-deprived the water. Ev values for processed water in the range of -200 millivolts (mv) or lower suggest an anaerobic environment. As can be seen in Figure 5, Ev values are generally below -200 mv with the exception of the positive value observed during June 1999. This value is likely due to the cell not processing water at the time of the site visit.

During operation, the pilot cell copper removal efficiencies were acceptable even when the cell surface was 40-percent covered in ice and the influent changed to an acidic water with a pH between 3.5 and 4. The pilot cell operated successfully (i.e., 89- to 97-percent removal of copper) for two years, after which time it was decommissioned.

B.4.6 Regulatory Challenges

The WYAMLP stated in their 2002 Annual Evaluation Summary Report that they had concerns that although the BCR cell may reduce contamination, it was unlikely to eliminate it. Because of current USEPA regulations, the WYAMLP was reluctant to assume the potential liability of becoming a principal responsible party under CERCLA for the mine discharge, should they make an unsuccessful attempt to correct the water quality problem.

B.4.7 Stakeholder Challenges

The primary stakeholder challenge associated with this site was negotiating the sale of sufficient land from the private property owner in order to construct a full scale treatment system. Ultimately, the WYAMLP was unsuccessful and the final system was never constructed.

B.4.8 Other Challenges and Lessons Learned

Minor temperature fluctuations may not affect SRBR overall activity all that much once the microbial suite has established itself. In the pilot scale SRBR at the Ferris-Haggerty Copper Mine/Osceola Tunnel in Wyoming, the mine effluent was typically less than 5°C. During the winter at this high altitude (9,500 feet elev.) site, the treated effluent temperature dropped to as low as 0.5°C. Yet the rate of sulfate reduction reduced to only about 80 percent of the 5°C benchmark rate (Gusek, 2000). This observation was incorporated into the design of a full-scale system to treat as much as 600 gpm.

A part of the temperature challenge at the Ferris-Haggerty site was mitigated by a number of design features, including:

- collecting the mine water deeper in the mine and delivering it to the SRBR cells in insulated pipes to preserve some of its ambient in-ground temperature, and
- covering the SRBR's with an insulating layer of clean mine waste rock.

Covering/burying SRBR's (which do not need plants or sunshine to function) can be used to solve other full scale design challenges. Other design considerations, such as seasonal changes in metal loading rates, may further mitigate temperature effects on SRBR performance.

The advantage of low maintenance requirements for passive treatment make it attractive compared to other more active technologies, especially at remote sites such as Ferris-Haggerty.

The WYAMLP stated in their 2002 Annual Evaluation Summary Report that it would have been prudent to negotiate land acquisition as a component of access for the treatment feasibility study.

B.4.9 References

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B.5 Fran Coal Mine Case Study

Clinton County, PA

The mining team would like to acknowledge David Heinze of ENVIRON International Corporation who submitted the Biochemical Reactors Case Study.

B.5.1 Site Information

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B.5.2 Name, Location, and Site Description

The Fran Mine Site is a 15 ha (37 acre) reclaimed surface coal mine in Clinton county, PA. The area was partially mined in the 1930's and additional coal recovery was undertaken between 1973 and 1977. Reportedly the "worst acid mine drainage (AMD) in the State of Pennsylvania", destroying nearly 6 miles of three otherwise pristine streams which are tributaries of the Susquehanna River (Gusek and Schueck 2004).[1] Buried pyrite-rich pit cleanings and tipple refuse were found to be producing severe AMD. The pyritic material is located in discrete piles or pods in the backfill. The pods and the resulting contaminant plumes were initially defined using geophysical techniques and confirmed by drilling. Isolating the pyritic material from water and oxygen will prevent AMD production (Schueck et al. 1996). The Fran Mine AMD is similar to acid rock drainage from metal mines in that it contains elevated concentrations of iron and aluminum as well as other heavy metals such as copper, zinc, cadmium, cobalt, chromium and nickel and has acidity on the order of grams per liter. Research has shown that AMD with these similar characteristics is amenable to a passive treatment technology that uses sulfate reducing bacteria. The sulfate reducing bacteria is

able to precipitate the heavy metals as sulfides and the aluminum as a hydroxy-sulfate mineral phase (Gusek and Schueck 2004)



Figure B.5-1. Fran Coal Mine MIW

The lower Kittanning coal seam was present in two splits separated by 10 to 20 feet of clay. Only the upper split was mined, leaving a thick underclay as pavement. The coal was overlain by black shale capped by a sandstone unit. The black shale is thought to be pyritic and acid producing. Infiltrating precipitation is the only source of groundwater. Acidic discharges developed soon after reclamation and were first noted after a fish kill in 1978. The discharges to surface and underground, estimated to average 35 gpm, destroyed five miles of native trout streams. The operator was unable to maintain treatment facilities and forfeited $9,400 in bonds.

B.5.3 MIW Chemistry

Buried pyrite-rich pit cleanings and tipple refuse were found to be the source of the Fran AMD in the pit backfill where the pyritic material was located in discrete piles or pods. These features and their resulting contaminant plumes were initially confirmed by drilling. The chemistry of the mine impacted waters (MIW) varies by season and is expressed below in Table B.5-1.

Table B.5-1. Spot analytical values

Date	January 2002	April 2003
Parameter	mg/L	mg/L
pH	2.3	2.3
Aluminum	343	221
Cobalt	2.7	0.8
Cadmium	0.11	<0.001
Copper	1.8	0.56

Date	January 2002	April 2003
Iron	792	185
Sulfate	4,963	2,140
Zinc	6.4	2.0
Calculated Acidity	4,370	2,060

Table B.5-2 lists pre-grouting water quality for the toe-of-spoil seep and selected monitoring wells which are considered to be representative of the site.[2]

The toe-of-spoil seep, labeled D3, and well FF62 represent the pre-grouting water quality which exits the site from the south and east lobes, respectively. Well FF62, located off site and adjacent to the east lobe, intercepts a subsurface discharge plume suspected to enter Rock Run as base flow. Also, well FF62 is located in the lower, unmined split of coal. The poor water quality demonstrates fracture communication between the pit floor and lower coal seam. D3 also appears to be coming from the lower coal seam. Comparison of the concentrations indicates that the water discharging from the east lobe (133) is of poorer quality than that discharging from the south lobe (FF62). Drainage from the majority of the pods of pyritic material flows towards the east lobe. Monitoring well L44 represents the pre-grouting water quality resulting from the Lower Kittanning spoil. L44 is not located downgradient of buried refuse or pit cleanings. Monitoring well L44 is considered a suitable control well to monitor normal water quality variations on the site because it is not influenced by the grouting activities. Flow from L44 would be toward the south lobe and eventually to the toe-of-spoil seep (Schueck et al. 1996).

Table B.5-2. Fran Mine water quality

	Data												
Mean Values	Monitoring Point D3 – Toe-of-spoil seep area												
	Lab	TDS	SO_4	Acid	Fe Total	Fe^{3+}	Al	Mn	Cd	Cu	Cr	Zn	Temp
	pH	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	µg/L	µg/L	µg/L	µg/L	Deg. C
	2.49	6,475	2,571	2,995	321	254	268	48.3	25.5	612	200	3.4	n/d
	Monitoring Well FF62 – East Lobe – Groundwater Discharge Point												
	2.32	7,970	3,477	4,088	876	737	256	39.2	83	806	221	4.3	12.0
	Monitoring Well L44 – Lower Kittanning Spoil												
	2.47	7,920	2,958	3,828	747	598	236	48.1	111	751	163	6.8	12.9
	Monitoring Well K23 – Severe AMD Production in Pods												
	2.1	46,352	15,639	21,315	5,437	1200	1,515	60.5	610	5,950	1108	27.6	13.3
	Monitoring Well X48 – Combined Spoil and Pyritic Pods												
	2.37	1,408	6,991	7,470	1,707	439	492	72.8	227	2,543	559	8.7	12.7

Monitoring well K23 demonstrates that the pods of refuse or pit cleanings can be sources of severe, localized AMD production. Concentrations of the mine drainage parameters in the water in and adjacent to these piles are often several times greater than the concentration of the same parameters in the discharge or elsewhere on the site. K23 is located within a pile of buried tipple refuse. It is located on the up-dip portion of the mine site and is also at the upper end of a pollution plume as defined by EM. The mean concentrations of the AMD parameters at K23 are four to six times greater than the concentrations of the same parameters in L44, which represents AMD generated by the spoil alone. This implies that enhanced AMD production from the pyritic material at K23 and similar locations significantly contribute to the degradation of the final discharge quality. The isolation of these subsurface pods of pyritic material from water and/or oxygen should improve the final discharge quality. This was the premise for this research effort.

Monitoring well X48 is located several hundred feet downgradient of piles of pyritic material, but is within the flow path of mine drainage as it migrates through the site towards the east lobe. The water quality sampled from this well would be influenced by both the Lower Kittanning spoil as well as AMD formed in the buried pods of pyritic materials. Note that the mean concentrations of the mine drainage parameters in well X48 are about double the mean concentrations of L44. X48 indicates dilution of the severe AMD as it migrates and mixes with less severe mine drainage. (Schueck et al. 1996)

Certain trace metals also contribute to the pollution from this site. Elevated concentrations of zinc, copper, chromium, cadmium, and arsenic were common in the drainage from this site. The concentrations of other trace metals, such as lead, nickel, and selenium were generally below detection limits

B.5.4 System Design

Because of the MIW chemistry, limestone-dominated vertical floor reactor cells were deemed inappropriate. This assumption was based primarily on the aluminum concentrations, vertical floor reactor systems plug with aluminum hydroxide (gibbsite) if the AMD contains more than 5-7 mg/L of dissolved aluminum unless a flushing feature is included in the design Gusek and Schueck 2004).

Properly designed sulfate reducing bioreactors do not suffer from the same limitations as vertical floor reactor systems. The SRBR design process typically involves bench scale tests to select the best organic substrate mixture followed by a longer pilot test to evaluate the concept under field conditions.

In 1992 and 1993, a grout injection program to isolate the pyritic material from water and oxygen and thus prevent AMD production was completed. The grout, composed of fluidized bed combustion (FBC) ash and water, was used in two different approaches that attempted pyrite isolation (Gusek and Schueck 2004).

Statistically significant water quality improvements have been noted as a result of the grouting, although resulted have varied. Any water quality improvements resulting from the grouting are

expected to be permanent because of the nature of the cementitious grout. However, the grouting program did not solve all AMD problems at the site. Residual AMD seepage from one zone of the backfilled pit continued to be problematic (Schueck et al. 1996).

A preliminary design for a full scale system used data from the winter of 2002-2003 which was reported as one of the wettest in recent memory in Pennsylvania. Anecdotal data suggest that the design peak flow rate for the full scale system would be about 30 gpm. Spot flow measurements in March 2003 revealed a flow rate of 82 gpm from the holding pond. A week later, the flow had reportedly dropped to 42 gpm. The size of the full scale system will be a function of assumed design flow rate. Absent more frequent flow data, it was assumed that the design flow will be 42 gpm and that any amount above that value will bypass the system and mix with the treated effluent with its buffering alkalinity. Flow rates about 42 gpm will be short lived, on the order of two weeks and that impacts to downstream locations will be mitigated by dilution with the elevated flows in the watershed in general.

The full scale system was planned to be comprised of the following components:

- A pipeline drain penetrating the pit boundary subcrop of the coal seam to divert AMD to the treatment system
- Four covered/buried sulfate reducing bioreactors plumbed in parallel
- An aerobic polishing cell discharging into an existing channel that feeds Camp Run.

Replacement of the organic substrate every 20 years was expected.

B.5.5 BCR Performance

On August 2001, five bench-scale SRBR cells were assembled on Pennsylvania State Forest maintenance facility on Cook's Run. The five various bench cell construction materials are summarized below:

- chipped wood (8-40 percent by weight)
- sawdust (10-40 percent)
- limestone (30 percent in all cells)
- alfalfa Hay (10 percent in all cells)
- cow manure (10 percent in all cells)
- cement kiln dust (1 percent in all cells, except #3)

Table B.5-3. Bench scale organic substrate mixtures

	Cell 1	Cell 2	Cell 3	Cell 4	Cell 5
Wood Chips	10%	25%	25%	40%	8%
Sawdust	40%	25%	25%	10%	21%
Limestone	29%	29%	30%	29%	50%

	Cell 1	Cell 2	Cell 3	Cell 4	Cell 5
Alfalfa Hay	10%	10%	10%	10%	10%
Cement Kiln Dust	1%	1%	0%	1%	1%
Manure	10%	10%	10%	10%	10%
Logic	High Saw-dust	Baselin-e	No Kiln Dust	High Wood Chips	High Lime-stone

The bench scale tests used 40 gallon capacity bench cells that were arranged in parallel to receive untreated Fran Mine water delivered by individual ISCO battery-powered automatic samplers, drawing form a common source tank that was replenished on a bi-weekly schedule. The cells operated for 18 weeks, then all cells were subjected to an autopsy to evaluate how much, if any, plugging occurred due to the precipitation of metal sulfides and other metallic compounds during the test period (Gusek and Schueck 2004).[1]

Prior to dissecting the bench cells, sample aliquots were recovered from the feed water holding tank and the discharge pipes and submitted for total metals analysis. The results are presented in Table 4.

Note the marked decrease in silicon, from about 77 mg/L in the feed water (average of 68 and 85 mg/L) to about 7 mg/L in the Cell 4 effluent. This reduction suggests that silicate-based metal compounds, perhaps higher density alumino-silicates, might be forming in the substrate in lieu of the typical gibbsite which is so problematic in many SAPS and other limestone-based passive treatment systems (Gusek and Schueck 2004).

Table B.5-4. ICP/AES bench results

Parameter	Feed	Bench 2	Bench 4	Bench 5
Aluminum	305	0.11	BDL	0.09
Cadmium	0.10	0.01	0.00	0.03
Cobalt	2.10	0.06	0.02	0.26
Chromium	0.16	BDL	BDL	0.01
Copper	1.61	BDL	0.01	BDL
Iron	709	38.10	0.66	94.21
Manganese	79	33	2.9	38
Nickel	2.93	0.03	0.04	0.28
Sulfate cal-culated	4,480	1,400	180	1,250
Silicon	67	11	7	42
Zinc	5.75	0.02	0.05	0.10

Based on metal removal data, it appeared that the substrate mixture in Cell 4 was slightly superior to the mixtures in Cells 1, 2, and 3, and clearly superior to the high limestone mixture in Cell 5 (Gusek and Schueck 2004).

The autopsies of the cells did not reveal any visible accumulations of aluminum precipitate, typically observed as a gel-like mass. The effluent pipe and limestone drainage layers were typically clean. Unfortunately, the cells were frozen, so it was not possible to closely examine the substrate for traces of aluminum deposits. Regardless, all cells operated from August to late December without any need for flushing to maintain flows. This is an advantage when looking from a maintenance standpoint at a remote mine location (Gusek and Schueck 2004).

Subsequently, PA DEP decided to install a 550-foot long underground drain to divert water away from the site. To neutralize the pH of any groundwater or rainwater that does flow through the site, 15,000 tons of alkaline material was to be packed into the ground. The agency planned to seed 27 acres, plant more than 15,000 trees, construct an access road and excavate and backfill 165,000 cubic yards of soil and material from the site (Spadoni 2012).

B.5.6 References

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B.6 Brewer Gold Mine Case Study

Chesterfield County, South Carolina

The mining team would like to acknowledge Jim Gusek, Sovereign Consulting, who submitted this Biochemical Reactors Case Study.

B.6.1 Site Information

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Name, Location, and Site Description

The Brewer Gold Mine site is located on the western border of Chesterfield County, South Carolina. The site is located in a rural area approximately one mile west of Jefferson, South Carolina. The property is bordered by Road 265 to the south, unoccupied land to the north and west, and Little Fork Creek to the east. The site is located near a topographic high, sloping to the east toward Little Fork Creek and to the west toward Lynches River. Little Fork Creek enters Lynches River approximately two miles downstream of the site.

Mining activities were conducted on approximately 230 acres of the site. Mining-influenced water (MIW) comes from two sources at the site: a flooded open pit and a spent cyanide heap leach pad. This case study focuses on the biochemical reactor (BCR) treating effluent from the spent cyanide heap leach pad (Pad 5), which was operated for an approximately 18 month test period in 1993 and 1994.

B.6.2 Mining-Influenced Water Chemistry

Influent water chemistry was monitored approximately weekly between September 1993 and December 1994. Field data (pH, Eh and conductivity) and samples for metals analysis (copper and iron) were collected routinely during this period. Manganese and aluminum were added to the routine sampling suite in October 1993. Sulfate and zinc were measured periodically throughout the test period. Pad 5 influent concentrations fluctuated in response to rainfall events on the heap and the presumed rise of pyrite oxidation activity as buffering leach solutions were rinsed out. Iron concentration varied from 8 mg/L after a leach solution flush to about 3,950 mg/L at decommissioning. Key metals concentrations and field data gathered during the Pad 5 Cell operation are summarized in Table B.6-1.

Table B.6-1. Average influent water chemistry, Brewer Mine, Pad 5 (9/93 – 12/94)

Parameter	Mean Concentration (Range)
pH	3.11 (2.06-5.92)
Eh	477 (223-611)
Conductivity	6985 (3590-8290)
Sulfate (mg/L)	3554 (2857-5200)
Copper (mg/L)	16.73 (2.23-102.4)
Zinc (mg/L)	0.12 (0.03-0.22)
Iron (mg/L)	318 (8-3160)
Manganese (mg/L)	0.68 (0.18-2.8)
Aluminum (mg/L)	(3.6-1.82)

B.6.3 System Design

The system designed to treat effluent from Pad 5 consisted of a single anaerobic pilot cell. A 2.4 ft. deep 4,000 ft^2 anaerobic cell was commissioned in early September 1993. The cell was filled with a mixture of composted turkey manure, sawdust, phosphate rock reject (limestone) and cow manure inoculum. Cell dimensions were determined by balancing the estimated volumetric sulfate reducing capacity (about 0.3 moles of sulfate reduced per day per cubic meter of substrate) with the molar loading of dissolved metals. The surface area was based on positive experience at other sites (800 ft^2 per gpm) with similar water chemistry. Although the cell was designed for a flow of 5 gpm, it actually treated 0.75 gpm for approximately 18 months.

B.6.4 BCR Performance

The BCR was operated by mine personnel, who were instructed to minimize ponding on the cell surface to preclude iron hydroxide formation. The effluent water chemistry varied considerably during the experiment. In one respect, this variation demonstrates the resilience of the treatment system. The Pad 5 cell was originally designed for a flow of 5 gpm. However, by the time the system was commissioned, influent water chemistry had deteriorated to the point that the flow was reduced to prevent overwhelming the cell's biochemistry.

Right after start-up, volunteer vegetation invaded the cell surface, marking the vegetation flourishes period of the cell history (Figure B.6-1). The cell vegetation was intentionally suppressed in two subsequent events and the cell refitted with a fresh source of native hay before the system was decommissioned in February 1995 to accommodate on-going reclamation/closure activities.



Figure B.6-1. Brewer Pad 5 biochemical reactor with vegetation

As shown in Figure B.6-2, after the start-up period until June 1994 (the vegetation flourishes period), the Pad 5 Cell had erratic metal removal performance. The redox/Eh during this time was also erratic, in concert with the metals removal with the exception of iron, which was nearly 99% removed. Copper removal was particularly variable; at times, copper in the effluent exceeded influent values. That is, copper was being remobilized. Aluminum removal paralleled copper values but never exceeded influent concentrations. Remarkably, effluent pH continued to be around 6.5 during this time despite large variations in influent pH due to leach solution being flushed from the heap after storm events or earthmoving activities.

The high redox/Eh values and other observations (e.g., effluent temperatures higher than 10°C) prompted the intentional elimination of the Pad 5 surface vegetation in June 1994, beginning the post-vegetation I period. Immediately thereafter and until the vegetation was again intentionally eliminated on October 27, 1994 (beginning of the post-vegetation II period), the percent aluminum, copper and iron removed became steadier and more consistently was in the range between 95% and 100%.

Influent metal concentrations began to increase at the beginning of the post-vegetation I period and increased steadily until decommissioning. Iron, in particular, increased to about 3,950 mg/; over an order of magnitude higher than the design concentration or the initial influent concentration. Remarkably, metals removal in the cell kept pace, especially during the post-vegetation II period as shown on Figure 5-1 for aluminum, iron and copper. Flow was maintained at about 0.75 gpm because it was difficult to adjust the flow any lower. Additionally, pH in the effluent decreased from 6.5 to 6.0 during the post-vegetation I period, which prompted the second intentional defoliation event. Towards the last month of the post vegetation I period, the percent removal of aluminum began to drop below 50%, corresponding to a drop in iron removal.

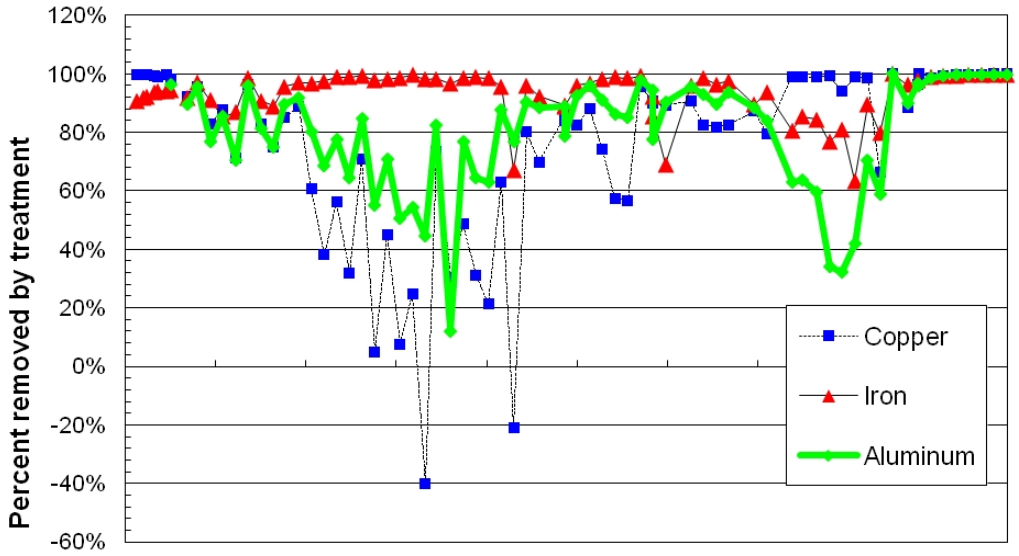


Figure B.6-2. Brewer Pad 5, combined total metals removal.

The vegetation was again eliminated on October 27, 1994. From this time to the decommissioning of the Pad 5 Cell, copper, iron and aluminum removal were consistently very near 100%. The effluent pH was observed to maintain a value of around 6.5. Sulfate influent and effluent differences increased even more markedly than the post vegetation I period.

B.6.5 BCR Monitoring

Influent and effluent samples were collected every few days during September 1993. Influent and effluent samples were collected approximately weekly from October 1993 through December 1994. All samples were analyzed for pH, Eh, conductivity, SO_4, copper, zinc, iron, manganese, aluminum and conductivity. Results are shown in Figure 5-1.

B.6.6 Regulatory Challenges

The primary regulatory challenge faced by this project was the variation in effluent water quality, discussed in detail in Sections 5 and 9 of this case study.

B.6.7 Stakeholder Challenges

The stakeholder challenges at this site are primarily those discussed in sections 5 and 9.

B.6.8 Other Challenges and Lessons Learned

After start-up, this cell apparently removed metals using oxidation reactions promoted by the plants on the cell surface; the limestone in the substrate probably assisted by buffering the iron hydrolysis reactions. This is supported by the excellent iron removal efficiency and the poor to negative removal efficiency for copper.

Once the vegetation was removed the first time, it is believed that anaerobic SRB-fostered reactions returned as the dominant removal mechanism. This is supported by the low redox/Eh, the marked improvement in metals removal, and the observed increases in sulfate reduction.

After the second defoliation, sulfate reduction exceeded design expectations which allowed the cell to continue to function despite the order of magnitude increase in metal loading, most notably iron. The sulfate reduction peaked at a rate of over 2.0 moles/day/cubic meter; it is suspected that this phenomenon would not have continued had the cell been operated for a few more months. The higher than expected rate was likely related to the readily digestible source of carbon provided by the recently killed vegetation. The root and stem tissues of the dead plants appeared to provide a superior nutrient carbon source compared to the hard wood cellulose (sawdust) already present in the substrate. The hay that had been added to the cell during retrofit probably functioned in the same manner. This is consistent with reports of others (Gusek 2000) using ethanol as an organic source.

A carbon mass balance analysis was conducted with the available data. During the period when vegetation flourished, the carbon supply to sulfate reducing bacteria was low because of the predominance of aerobic conditions in the substrate. After the successive defoliations, and as sulfate reduction became the predominant biogeochemical reaction in the cell, carbon utilization for sulfate reduction tracked each other remarkably well. That is, all carbon needed to allows sulfate reducers to produce sulfide for metals removal appeared to come from the hard wood substrate and the recently killed plants.

B.6.9 References

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B.7 West Fork Mine Case Study

Reynolds County, Missouri

The mining team would like to acknowledge James J. Gusek, Sovereign Consulting Inc. who submitted the Biochemical Reactors Case Study.

B.7.1 Site Information

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Name, Location, and Site Description

The West Fork Mine is an operating lead-zinc mine near Centerville, Missouri located on the West Fork of the Black River. The mine was initially developed and operated by ASARCO, Inc., and sold to The Doe Run Company in 1998. The mine produces about 46,000 tons of lead, 6,800 tons of zinc and 3,900 kilograms of silver per year.

Water drainage from the mine is discharged to the West Fork under a National Pollution Discharge Elimination System (NPDES) permit. The mine discharges about 1200 gallons per minute (gpm), which makes up approximately 10% of the total flow in West Fork. The discharge limits in the NPDES permit are based upon water-quality based requirements.

The West Fork Biotreatment Project won a Pollution Prevention Environmental Excellence Award from the US Environmental Protection Agency in 1995. It also won the 1998 Engineering Excellence Award in Wastewater Treatment – Colorado Section of American Consulting Engineers Council (ACEC): National ACEC Honor Award.

B.7.2 MIW Chemistry

This operating underground lead mine has a neutral pH discharge with 0.4 milligrams per Liter (mg/L) lead and 0.18 mg/L zinc; flow is about 1200 gpm.

B.7.3 System Design

The large scale system was designed based on the performance of a pilot scale system and interim bench scale studies. The large scale system cost approximately $700,000 (including engineering and permitting) and required about four months to construct in 1996. System operational costs

include water quality monitoring as mandated by law. No additional costs for reagents are incurred; since the system uses gravity flow, moving parts are few and include valves, minor flow controls and monitoring devices. Based on carbon depletion rates observed in a pilot system, the anaerobic cell substrate life was projected to be greater than 30 years. Should mine water quality deteriorate, the full scale design included a 50 percent safety factor. The pilot scale system (25 gpm) was tested by operating for about 90 days at double the design capacity; compliant effluent with respect to total lead concentration and other key performance parameters resulted from this test.

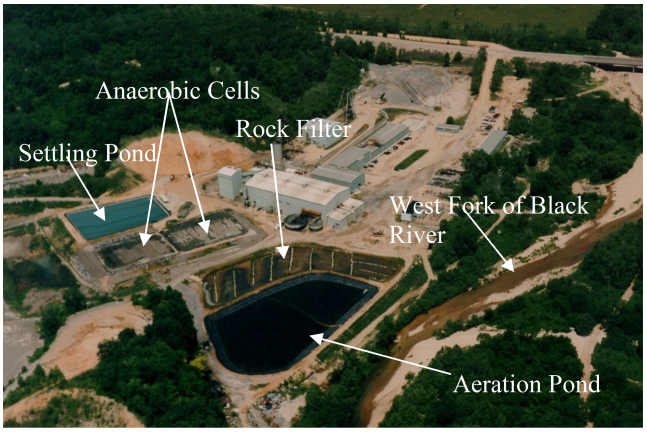


Figure B.7-1. Aerial photo of West Fork mine passive treatment system, Missouri

B.7.4 BCR Design

The passive treatment system is composed of five major parts: a settling pond, two anaerobic units (BCRs), a rock filter, and an aeration pond. The system is fully lined. The design was also integrated into the mine's pre-existing fluid management system. Specific design characteristics include:

- A rectangular-shaped, 40 mil HDPE-lined settling pond with a top surface area of 32,626 square feet (0.75 acres) and a bottom surface area of 20,762 square feet (0.48 acres). The sides have slopes of 2 horizontal to 1 vertical (2H:1V). The settling pond is nominally 10 feet deep. It discharges through valves and Parshall flumes into the two anaerobic units.
- Two anaerobic BCR units are used, each with a total bottom area of about 14,935 square feet (0.34 acres) and a top area of about 20,600 square feet (0.47 acres). Each unit is lined with 40 mil HDPE and was fitted with four sets of fluid distribution pipes and three sets of fluid collection pipes, which were subsequently modified. The distribution/collection pipes were connected to commonly-shared layers of perforated HDPE pipe and geonet materials sandwiched between layers of geofabric. This feature of the design was intended to allow management of sulfide production in hot weather by decreasing the retention time in the unit through intentional short circuiting. The spaces between the fluid distribution layers were filled with a mixture of composted cow manure, sawdust, inert limestone, and alfalfa. The total thickness of substrate, piping, geonet, and geofabric was about six feet. The surface of the anaerobic units was covered with a layer of crushed limestone. Water treated in the anaer-

obic units flows by gravity to a compartmentalized concrete mixing vault and thereafter to a rock filter unit. The gravity-driven flows can be directed upward or downward.

- The rock filter is an internally bermed, clay-lined shallow unit with a bottom area of about 63,000 square feet (1.4 acres) and a nominal depth of one foot. It is constructed on compacted fill that was systematically placed on the west side of a pre-existing mine water settling pond. Limestone cobbles line the bottom of the unit and the unit is compartmentalized by limestone cobble berms. It was planted with cattails (*Typha latifolia*) and inoculated with local species of algae.
- The discharge from the rock filter flows through a drop pipe spillway and buried pipe into a 40 mil HDPE lined aeration pond. The aeration pond surface covers approximately 85,920 square feet (2.0 acres). The aeration pond discharges through twin 12-inch HDPE pipes into a short channel that leads to monitoring outfall 001 and thence into the West Fork of the Black River.

Bench-scale test results suggested that the anaerobic units be incubated with settled mine water for about 36 hours or less before fresh mine water was introduced at full flow to minimize initial levels of biological oxygen demand (BOD), fecal coliforms, color, and manganese. For about two weeks, pumps recycled the water within the two BCR units. Based on data collected in the field, and subsequent laboratory confirmation, the water from the BCR units was routed to the tailings pond for temporary storage and later treatment and release. At that point, the rock filter and aeration ponds were brought on-line. In the meantime, the mine discharged according to plan through an overflow pipe from the settling pond as it had during construction of the other components.

After about six weeks of full-scale operation, the apparent permeability of the substrate was found to be lower than expected and the system was operating nearly at capacity. The system had been designed so that either of the two BCR units could accept the full flow amount on a temporary basis in case maintenance work required a complete unit shutdown.

B.7.5 BCR Performance

The average influent water quality can be compared with discharge water quality during the June through November 1997 period. Discharge levels of lead and other metals were reduced substantially from average influent levels. For lead, the level was reduced from a typical average of 0.40 mg/L to between 0.027 and 0.050 mg/L. Zinc, cadmium, and copper effluent concentrations were also reduced.

Table B.7-1. BCR performance data

Parameter	Typical Average Influent (mg/L)	Effluent Range June-Nov 1997 (mg/L)
Lead (Pb)	0.4	0.0270 – 0.050
Zinc (Zn)	0.36	0.055 – 0.088

Parameter	Typical Average Influent (mg/L)	Effluent Range June-Nov 1997 (mg/L)
Cadmium (Cd)	0.003	<0.002
Copper (Cu)	0.037	<0.008
Oil and Grease	-	<5.0
Hydrogen Sulfide (H_2S)	-	0.0110 – 0.025
Phosphorus, total	-	<0.05 – 0.058
Ammonia, as N	0.52	<0.050 – 0.37
Nitrate and Nitrite	2	<0.050 – 1.7
True Color	-	10 – 15
BOD	1.7	<1 – 3
Fecal Coliform	-	<1 – 2
pH	7.94	6.63 – 7.77
Total Suspended Solids (TSS)	-	<1 – 4.2

Of the five parts of the system, the operation of the rock filter has been the most interesting. It operates as a natural wetland where water of a depth of 1 to 2 feet (30 to 60 cm) meanders through the limestone cobbles. Flora and fauna have thrived in this ecosystem. It has served the important function of cleansing the excess sulfide in the water that is leaving the anaerobic units. From July 1997 to September 1998, the average of 55 analyses of sulfide concentration in the water entering the rock filter was 3.3 mg/L. In 55 analyses of sulfide in the rock filter effluent, sulfide was detected in the water 20 times and none of these were above 0.25 mg/L.

Because the water entering the rock filter contains a significant concentration of sulfide, a unique ecosystem of algae and bacteria have developed in this area. In the summer of 1997, red algae/bacteria started to develop in this influent area and persisted. In addition, a white, soupy scum developed in this area. During the summer of 1997, when high levels of sulfide were entering the rock filter, the water would develop a milky white colloidal suspension that would persist throughout the wetland system. This milky suspension had diurnal characteristics; it was more persistent in the morning and sometimes clear up during the day. In the summer of 1998, this milky suspension was not as evident even though the concentrations of sulfide entering the rock filter were sometimes higher. Vegetation in the rock filter was much lusher in the second summer. The speculation is that this milky suspension was colloidal sulfur, which then was removed by this form of wetland ecosystem.

Besides removing sulfide from the water, the rock filter also plays a significant role in further reducing the concentration of lead in the water. Over four seasons from July 1997 to July 1998, the average concentration of 40 analyses of total lead in the water entering the rock filter was 0.085 mg/L and the average concentration of lead in the water exiting the rock filter was 0.050 mg/L. The mechanism for lead removal in the rock filter was not known, but is suspected to be adsorption to MnO_2.

B.7.6 BCR Monitoring

None reported.

B.7.7 Regulatory Challenges

Although discussions about using a passive treatment system had been held previously, the system's original permit application was made to Missouri Department of Natural Resources after the pilot scale test had been operating for one year and on-going data from the test was added during the permit review process. This allowed regulators to be assured that an organic-based wetland-type substrate could remove dissolved lead from mine effluent as there had been some concerns regarding how effective the system would be.

The use of cow manure as an ingredient in the anaerobic cell substrates was a special regulatory hurdle because its use raised issues of BOD, fecal coliform bacteria and other organic-related water quality criteria problems from a non-degradation of West Fork perspective (Gusek 1998). Additional testing was requested by Missouri DNR, including monitoring for fecal coliform, color, BOD, and other minor constituents.

DNR was also concerned about the closure and reclamation of the biotreatment system at the end of the West Fork facility life. The system was constructed within the boundaries of the waste management areas as defined by the Metallic Minerals Waste Management Act and was, by definition, a waste management structure. Therefore, closure and reclamation activities would adhere to Section 5 of the Metallic Minerals Waste Management Permit issued to ASARCO's West Fork Unit in January, 1991(Gusek 1998).

B.7.8 Stakeholder Challenges

None observed.

B.7.9 Other Challenges and Lessons Learned

B.7.9.1 Hydrogen Sulfide Gas Lock

Research found that H_2S gas, generated by the sulfate-reducing bacteria, was being retained in the substrate in the anaerobic units; this created a gas-lock situation that prevented full design flow. A temporary solution was obtained by periodic burping of the units using the control valves. However, the burping had to be performed at 24-hour intervals, which was determined that this solution was too labor-intensive.

The sulfide gas lock problem was investigated in December 1996 by installing vent wells in the substrate and measuring the gas pressures. Observations indicated that the gas was a factor in apparent short circuiting of the water passing through the unit. The layered geotextiles (geonet and geofabric), originally intended to promote horizontal flow, appeared to be trapping the sulfide gas beneath them and vertical flow was being restricted. The permeability of the substrate itself was for

the most part unaffected. However, construction practices in the south BCR unit could have contributed to the situation. Here, a low ground bearing bulldozer was used to place substrate in nominal 6-inch (15-cm) lifts. This could have created a layering effect that may have trapped gas as well. Substrate layers in the north anaerobic unit were placed in a single lift, and no layering effect was observed during subsequent excavation. It is noteworthy that the mid-cell geotextiles had not been a feature of the pilot test unit design.

The first phase of a permanent solution was implemented with a trenching machine that ripped through the geonet/geofabric layers in the south anaerobic unit. This disrupted the gas-trapping situation. Subsequently, the substrate from the entire south anaerobic unit was excavated and the unit refilled without the geotextiles in June 1997. Identical action was taken on the north anaerobic unit in September 1997. These actions have solved the gas lock problem.

B.7.9.2 Changes in Design from Pilot to Full—Scale System

A properly designed passive treatment system should be based on a phased testing program of laboratory-, bench-, and pilot-scale experiments. These experiments and the subsequent design must take into account the physical availability of some construction materials. Bench testing may have identified a superior type of organic component that the SRB favored, but it may not be available in sufficient quantities to warrant including it in the final design. Local farmers in particular are notorious for offering to give away animal manure during the testing phase of the project only to boost the price to capitalize on a captive market when large quantities need to be procured. Contractors and project owners seek relief from these situations by substituting similar but less expensive sources, which are virtually the same. Again, the West Fork Project in Missouri provides a couple of instances where minor digression from the pilot design caused subsequent problems in the full-scale system.

Gusek (2000) reported that to allow better flow control/system throttling in the full-scale BCR units during the summer, intermediate layers of perforated pipes were installed in the substrate at the 2-foot and 4-foot depths. To facilitate water collection/dispersion, the pipes were sandwiched between a layer of geonet and two layers of geotextile. Due to project scheduling, there was not time to test this concept on a pilot scale; the design change appeared to be minor. Another minor design change occurred during construction of the full-scale system. Alfalfa hay that was used in the construction of the pilot was in short supply; a source of spoiled alfalfa pellets was offered as a substitute and approved by the field engineer.

The two combined changes above had significant impacts on the ultimate hydraulic performance of the BCR units. While the geochemical characteristics of the substrate mix met the design specifications, the physical situation caused by the changes was a significant departure from the pilot design. First, the geotextile trapped some of the gases evolved from the biological activity and created a gas-lock condition that restricted fluid flow through the unit. Second, the substitution of the alfalfa product in place of the baled source yielded a substrate with a slightly lower saturated permeability than that measured in the pilot. The net result was a system that was geochemically sized to temporarily treat elevated flows, but the flow restrictions prevented this design feature of the

system from being used. The condition was ultimately fixed, but a valuable lesson was learned. Even minor deviations from bench- or pilot-scale configurations or design can result in major changes in system performance and should be avoided as much as possible.

B.7.9.3 Clogging

It was expected that the BCR system would be virtually maintenance free which was not the case with the anaerobic units. Keeping these units from clogging has required periodic rototilling and back-flushing, and after almost two decades, complete reconstruction. As of 2012, the BCR units are to be rebuilt due to clogging with rock flour from active mine operations. Because attempts were made during the summer to use only a portion of the two units, maintenance has been more extensive at this time than during the winter. Nevertheless, these units have performed according to design and have been effective at removing lead from the mine water for almost two decades, albeit with several major renovations. Because of this necessary maintenance, the design of the plumbing system to include back-flushing, upflow and downflow, and use of only a portion of the unit has been particularly advantageous.

B.7.10 References

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B.8 Leviathan Mine Case Study

Alpine County, California

Acknowledgements

The mining team would like to acknowledge Dr. Nirupma Suryavanshi, Cal EPA, who submitted this Biochemical Reactors Case Study.

B.8.1 Site Information

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Name, Location, and Site Description

The Leviathan Mine is the former site of intermittent mining operations dating back to the 1860s, and open pit sulfur mining operations from the 1950s through the 1960s. A pilot scale bioreactor was first installed at the Leviathan Creek seep in 1993, and transferred to the Aspen Creek seep in the late 1990s. In 2003, Atlantic Richfield Company and researchers from the University of Nevada-Reno (UNR) and the U.S. Environmental Protection Agency (USEPA) installed a full-scale compost-free sulfate reducing bioreactor system (SRB) to treat acid rock drainage (ARD). Over an evaluation period of 20 months, from late 2003 to summer 2005, the bioreactor was able to achieve a target-metal removal efficiency of 95 percent. All target metals, except iron, were reduced to concentrations below the USEPA interim discharge standards. The compost-free bioreactor system also raised the pH of the ARD from 3.0 to 7.0 and treated influent flows up to 30 gallons per minute (gpm) year-round. This case study looks at the effectiveness of the sulfate-reducing bioreactor treating ARD from the Aspen Seep at Leviathan Mine.

The Leviathan Mine is located in Alpine County, California near the California-Nevada border. The disturbed land comprises approximately 250 acres at the 7,000-foot elevation on the eastern slope of the Sierra Nevada. Mining operations commenced in the 1860s, but the mine was inactive from 1872 to 1935. The mine operated intermittently until the Anaconda Company purchased the property in 1951 and extracted sulfur by open pit mining from 1952 to 1962. No significant mining activities have occurred since Anaconda ceased operations in 1962 and sold the property.

Major environmental damage occurred at the mine, which is surrounded by the Humboldt-Toiyabe National Forest, during the period of open pit mining. Snowmelt, rain, and groundwater interact with the waste rock, creating sulfuric acid, which in turn leaches additional contaminants from the

native minerals such as arsenic, copper, cadmium, nickel, zinc, chromium, aluminum, selenium and iron. Manganese is a secondary contaminant. The resulting ARD flows into the Leviathan Creek system at numerous points, eventually joining the East Fork of the Carson River. For most of the year, roughly half of the flow in Leviathan Creek is composed of ARD. The affected media include soil, sediment, surface water (stream, rivers, runoff, and drainage), surface pool water (lakes, ponds, and pools), and groundwater.

Leviathan Mine was added to the National Priority List (NPL) in May 2000 to address contamination of surface water from acid mine drainage (AMD) and ARD. USEPA identified two problems requiring immediate attention: (1) an evaporation pond, known as Adit Drain (AD), collecting highly contaminated acid drainage, which overflows into the Leviathan Creek during the spring snowmelt; and (2) three seeps of acidic drainage causing contamination to enter Leviathan Creek and Aspen Creek (Figure B.8-1). One of these seeps, Aspen Seep (AS), originates from dumped overburden and flows into Aspen Creek. A sulfate-reducing bioreactor (SRB) designed and operated by Atlantic



Figure B.8-1. Leviathan mine disturbed area with major known ARD points including Aspen Seep.

Richfield Company and University of Nevada-Reno (UNR) and USEPA researchers, treats the drainage from the Aspen Seep. More traditional lime-based treatment systems are currently being used to treat the other two contamination sources (Delta Seep/Channel Underdrain and Adit Drain) at the Leviathan Mine site. The overflow water is treated by chemical precipitation to mitigate human health and ecological risks. The full-scale treatment process, used seasonally since 2005, includes lime neutralization, aeration, and oxidation of reducing metals with the rotating cylinder treatment system (RCTS). The quantity being treated ranges between 3 million and 20 million gallons annually. The final, long-term remedy for this site has not been selected.



Figure B.8-2. Aerial view of the Leviathan Mine.

B.8.2 MIW chemistry

ARD released from the Aspen Seep into Aspen Creek contains elevated levels of four primary metals: aluminum, copper, iron, and nickel. Each of these metals has historically exceeded USEPA interim discharge standards by over 500 times. Secondary metals of concern include selenium and zinc. Fish and insect kills in Leviathan Creek, Bryant Creek, and the East Fork of the Carson River have been attributed to the release of metal-laden ARD. ARD, at pH 3, flows from the Aspen Seep at rates ranging from 8 to 30 gallons per minute (gpm). Table B.8-1 shows average concentrations of these primary metals, the pH prior to treatment, and compares the concentrations to the USEPA Interim Discharge Standard for Leviathan Mine.

Table B.8-1. Discharge criteria

Water Quality Parameter	Maximum[2]	Average[4]
pH [1]	Between 6.0 and 9.0 SU	
Arsenic (dissolved)	0.34 mg/l	0.15 mg/l [3]
Aluminum (dissolved)	4.0 mg/l	2.0 mg/l
Cadmium (dissolved)	0.009 mg/l	0.004 mg/l
Chromium (dissolved)	0.97 mg/l	0.31 mg/l
Copper (dissolved)	0.026 mg/l	0.016 mg/l
Iron (dissolved)	2.0 mg/l	1.0 mg/l
Lead (dissolved)	0.136 mg/l	0.005 mg/l
Nickel (dissolved)	0.84 mg/l	0.094 mg/l
Selenium (total recoverable)	Not Promulgated	0.005 mg/l
Zinc (dissolved)	0.21 mg/l	0.21 mg/l

1. pH measurement based on 24_hour (single day) average discharge.
2. Concentrations based on daily grab samples, each grab sample field-filtered and acid fixed promptly after collection.
3 All average concentrations based on four daily grab samples, each grab sample field-filtered and acid fixed promptly after collection.
4 If the concentration detected by the contract laboratory is less than the detection limit, detection limit is used in calculating the average concentration.

USEPA 2008 Discharge Criteria in Table 1 are based on Applicable or relevant and appropriate requirements. Performance criteria include measuring the contaminant concentrations in the effluent water discharged to Leviathan Creek to ensure compliance with Environmental Protection Agency and state of California regulatory objectives.

B.8.3 BCR Design

The State of California, the site owner and therefore partially responsible for cleanup, had funded a bioreactor treatment system at the Leviathan Mine since the early 1990s. The system started as a simple one-cell, pilot-scale bioreactor with a manure substrate. The system continued to evolve throughout the late-90s but did not take its current form until the site was listed on the NPL. After the site became final on the NPL, USEPA directed Atlantic Richfield to prevent ARD discharge from the Aspen Seep and several other discharge points. The State of California wanted the bioreactor system to be part of the Atlantic Richfield responsibility for cleanup. Atlantic Richfield saw

promise in the system but felt it needed some improvements. The USEPA Office of Research and Development (ORD), Atlantic Richfield, the State, and UNR researchers convened for a design session to offer different approaches and ideas for constructing and implementing a full-scale bioreactor treatment system at the site. As a team, they proposed a system design, and the full-scale, compost-free bioreactor system was constructed in 2003. As constructed, the system requires 0.75 acres.

The bioreactor at the Leviathan Mine Aspen Seep relies on sulfate-reducing microbial organisms, such as *Desulfovibrio* sp., to reduce sulfate to sulfide. These organisms function at a critical pH 4.0 (Tsukamoto and Miller, 2005). ARD from the Aspen Seep has pH 3.1 and, therefore, requires pretreatment before entering the compost-free bioreactor treatment system. In order to accommodate this requirement, a 25 percent sodium hydroxide solution (0.26 ml/L) is added to the influent in a pretreatment pond (1,000 ft^3). The influent is effectively increased to pH 4.0 before it enters the compost-free bioreactor system. Ethanol (0.43 ml/L) is also added to the system to provide a carbon source for the sulfate-reducing microbes.



Figure B.8-3. Aspen Seep Bioreactor No. 2 lined with HDPE and filled with river rock.

After addition of the sodium hydroxide solution and ethanol, ARD flows to Bioreactor No. 1 to reduce sulfate to sulfide. Bioreactor No. 1 measures 12,500 ft^3 in total volume and 5,300 ft^3 in active volume, with a 22-hour hydraulic residence time (HRT) at 30 gpm. The bioreactor is lined in 60 mil high-density polyethylene (HDPE) and is filled with 6- to 24-inch river rock. Along with supplying a substrate for the bacteria to grow on, the river rock also provides stable flow paths and allows precipitates to be flushed through the matrix. Sulfide generated in the first bioreactor is

passed to the second bioreactor for additional metals removal. With a 13-hour HRT at 30 gpm, Bioreactor No. 2 measures 7,000 ft^3 in total volume and 3,000 ft^3 in active volume (Figure3). Each bioreactor has three influent distribution lines and three effluent collection lines at different elevations to allow variable flow operations.

After passing through the bioreactors, a 25 percent sodium hydroxide solution is once again added to the effluent to increase the pH to a neutral condition. A continuous-flow pond, measuring 16,400 ft^3 with a 68-hour HRT at 30 gpm, collects the effluent from the second bioreactor for extended settling of metal sulfide precipitates. The effluent from this settling pond flows over a rock-lined aeration channel, measuring 150 feet long and two feet wide, to promote degassing of residual hydrogen sulfide prior to discharge.

Chemical reaction (addresses the citation in section 3.1 Alcohol bases) for sulfate-reducing bacteria using an alcohol carbon source:

$$4AH^2 + SO_4^{2-} + H+ = 4A^{2-} + HS+ 4H_2O$$

$$H_2S + M^{2+} = MS + 2H^+$$

AH2 is the carbon source and SO_4^{2-} is the terminal electron acceptor in the electron transport chain of the sulfate-reducing bacteria. This causes an increase in pH. H2S reacts with metals and results in metal sulfide precipitate (MS).

The reduction of sulfate to sulfide:

$$H_2SO_4 + 8H^+ + 8e^- = H_2S + 4H_2O$$

Ethanol contributes 12 electrons per molecule oxidized.

$$3H_2O + CH_3OH \rightarrow 12e^- + 2CO_2 + 12H^+$$

Electron counting enables determination of the amount of carbon source required to reduce sulfate

To prevent plugging of the rock matrix, precipitate slurry is flushed occasionally from the bioreactors. The slurry is settled in a flushing pond (18,000 ft^3, 75-hour HRT at 30 gpm). Occasionally, solids are pumped out of the settling and flushing ponds and dewatered using a 10- to 15-foot spun-fabric bag filter. Under California and Federal standards, the bag filter solids are not hazardous.

The total system HRT is 107 hours at maximum design flow of 30 gpm.

B.8.3.1 Modifications to initial design

Although the system has experienced nearly constant tweaking, the following is a description of the three major bioreactor treatment system designs: manure substrate pilot-scale; two-cell bioreactor, and full-scale compost-free.

The original pilot-scale bioreactor developed by the UNR research team in the early 1990s was a simple system consisting of a manure substrate in a small, shallow pond (Tsukamoto and Miller, 2005). This organic substrate served as both the physical structure and the sole carbon source for the sulfate-reducing bacteria. However, once the carbon source was depleted, ARD treatment slowed down. Additionally, ARD from the Leviathan Seep proved too acidic for the sulfate-reducing bacteria to function optimally, further reducing the efficiency of waste stream treatment. The drainage at Leviathan Creek was from waste rock, described as material containing less than 20 percent sulfur by mass. After the first year of operation, the researchers determined the bioreactor was ineffective in treating ARD from the Leviathan Seep.

In order to address problems with the initial bioreactor design, the system experienced a facelift in 1998 (Tsukamoto and Miller, 2005). A new two-cell bioreactor was constructed at the Leviathan Mine Aspen Seep. The ARD from the Aspen Seep, originating from dumped overburden, is a less acidic waste stream and enabled the microbes to have a better chance of survival. In addition to the location, key modifications implemented in the 1998 bioreactor included:

- developing a two-cell system utilizing wood chips and an inert rock matrix
- employing alcohol as a carbon source
- adding base to further increase the alkalinity of the ARD
- allowing precipitates to be flushed from the bioreactor cells

The use of alcohol as a carbon source provides an advantage over other organic substrates. Alcohol can be used for treatment over extended time periods, and it also maintains a liquid state under varying environmental temperatures. Finally, the addition of alcohol to a treatment system can be varied according to optimal operating conditions. Alcohol, specifically ethanol, is an ideal substrate for use at the Leviathan Mine Aspen Seep due to the remote nature of the site as it is only accessible for a few months out of the year (see Section 3.1 regarding influent variability and the advantages of liquid substrates, and also Section 6.2.3 regarding seasonal access).

Again in 2003, the system was redesigned and the compost-free bioreactor treatment system was constructed. The most recent design of the bioreactor uses a rock matrix in both bioreactor cells, includes a pretreatment pond, and has improved flow distribution and advanced sludge capture capability.

B.8.3.2 Operation mode: gravity-flow vs. recirculation

Over the first six months of evaluation, the 2003 bioreactor design operated under gravity-flow mode. Gravity-flow mode allowed metal precipitates to accumulate in both the bioreactors and the settling pond. This required the system operators to flush the system frequently, in turn disturbing the bacteria in the bioreactors. In order to avoid the need to flush the system so frequently, the researchers transferred the system into recirculation mode for the remaining 14 months of the evaluation period. In recirculation mode, untreated ARD is mixed with a 25 percent sodium hydroxide solution and sulfide-rich water from Bioreactor No. 2. The mixture then flows into the settling pond where the high pH and high sulfide concentrations encourage precipitation of metal sulfides.

This prevents the metals sulfides from precipitating in the bioreactors. The pH of the water moving through the bioreactors is nearly neutral, presenting ideal conditions for the sulfate-reducing bacteria. The system required 17 percent less sodium hydroxide while operating under recirculation mode.

B.8.4 BCR Performance

The 2003 compost-free bioreactor was evaluated between November 2003 and July 2005 as part of the Superfund Innovative Technology Evaluation (SITE) program. This effort was possible through the cooperation of the USEPA National Risk Management Research Laboratory (NRMRL), USEPA Region IX, the State of California, Atlantic Richfield, and UNR.

Table B.8-2. Bioreactor Treatment System Removal, Full-scale Compost-free Bioreactor

	Gravity-flow Mode (11/2003 – 4/2004)		Recirculation Mode (5/2004 – 7/2005)		USEPA Interim Discharge Standard (mg/L)
	Influent (mg/L)	Effluent (mg/L)	Influent (mg/L)	Effluent (mg/L)	
pH	3.1	7.2	2.9	7.6	-
Al	37.5	0.1	40	0.05	2.0
Fe	117	4.9	116	2.7	1.0
Ni	0.49	0.07	0.53	0.07	0.094
Cu	0.69	0.005	0.79	0.005	0.016
Sulfate	1502	1222	1530	1170	-

The system achieved an increase in waste stream pH from 3.0 to over 7.0 during treatment. Additionally, although the influent concentrations of the target metals were up to 580-fold greater than USEPA interim standards, effluent concentrations were up to 43-fold below the standards. The system was also able to reduce the sulfate concentration in ARD by more than 17 percent.

During the first six months of evaluation, while the system operated in gravity-flow mode, 2.44 million gallons of ARD was treated. The system used 2,440 gallons of sodium hydroxide solution and 1,180 gallons of ethanol. Removal efficiency of target metals exceeded 94 percent.

Over the following 14 months, while the system functioned in recirculation mode, over 5.8 million gallons of ARD was treated achieving a removal efficiency of target metals exceeding 96 percent. During this time, 5,280 gallons of sodium hydroxide and 2,805 gallons of ethanol were pumped into the system. The system operates year-round and treats up to 30 gpm in either mode.

The innovative two-phase lime treatment system to treat millions of gallons of the worst acid discharge that collects throughout the year in several large ponds on the mine site was developed. The treatment occurs during the summer months to maximize the pond storage capacity and prevent overflow of the ponds in the winter and spring. The California Regional Water Quality Control Board - Lahontan Region - continues to capture the highly acidic waste from the Adit Seep (AS) and Pit Under-drain (PUD) in the existing pond system and then treat the entire year's

accumulation during an intensive one- to two-month period during the summer. Pond overflow, initially considered the most serious contaminant source, has been prevented since 1999. In the wet year of 2006, nearly 20 million gallons were treated during the spring and summer. In very dry years such as 2007 and 2009, only three million gallons or less must be treated to completely empty the ponds. Improvements in equipment, process efficiency and monitoring were constantly being implemented.

Atlantic Richfield Company, the successor to Anaconda, was capturing and neutralizing most of the remaining acid rock drainage as they develop long-term cleanup plans. A full-scale biological treatment system designed by the University of Nevada-Reno and constructed and operated by Atlantic Richfield has gained international attention for its remarkable success in treating one of the seeps year-round. Since 2003, the biological treatment system captured and treated all the acidic drainage captured at the Aspen Seep, typically more than three million gallons per year.

Atlantic Richfield has used various lime treatment systems to treat approximately six million gallons of acid mine drainage annually from the Delta Seep (DS) and Channel Underdrain (CUD), but only during the summer months.

Channel Underdrain (CUD), Delta Seep (DS) and Aspen Seep (AS) were included in the system. Atlantic Richfield Company attempted to design a system to treat all the known discharge sources for the entire year to reduce environmental damage, allow a more thorough evaluation of remaining risks to the downstream area and to assess the effectiveness and reliability of potential long-term treatment options. Difficulties were encountered during 2006 indicated that a lime treatment plant is not likely to be effective during the winter without very large investments in safe access and power at this remote mountain location. Improvements in the temporary lime treatment and biological systems, including a state-of-the-art High Density Sludge (HDS) lime treatment plant, were completed in 2009. The systems have improved efficiency with less waste generated and were also allowing an earlier commencement of the treatment season for the Channel Under-drain and Delta Seep. It was expected that the decision-making and design processes will continue for the next several years before a proposed plan for a long-term, year-round remedy is developed for public comment.

Atlantic Richfield's contractor AMEC continued to operate the Aspen Seep Bioreactor (ASBR) in recirculation mode to treat AS discharge without significant modifications from 2010 operations. A belt filter press was used to dewater the ASBR sludge after AMEC determined a centrifuge mobilized to the site earlier in the year was not appropriate for dewatering ASBR sludge. The dewatering system consisted of a belt filter press, associated tanks for containment of raw sludge and dewatering effluent, and sludge and effluent pipelines.

When the ASBR operates in recirculation mode, sodium hydroxide is added to the AS influent which is then mixed with sulfide-rich water and sodium hydroxide from bioreactor cell 2 in the Settling Pond (Pond 3) for precipitation of metal sulfides. Water is pumped from the opposite end of the Settling Pond back to the pretreatment pond at approximately three times the system influent rate. The ASBR is operated in this mode because recirculated water has lower metals content and higher pH than raw AS water. These parameters are ideal for biological treatment in the bioreactor

cells. Ethanol is added to the recirculated water to provide a carbon source for the sulfate-reducing bacteria. The Settling Pond (Pond 3) discharges to the next Settling Pond (Pond 4) from which bioreactor effluent is discharged along a rock-lined aeration channel to an infiltration pond adjacent to and above Aspen Creek. The recirculation pumps are powered by propane generators.

Approximately 7.2 million gallons of water from the AS bioreactor were treated and discharged to Aspen Creek during 2011. The AS flow rate during 2011 was higher than the 2010 AS flow rate. Details of the bioreactor performance are summarized in Atlantic Richfield's year-end summary report.

Field parameter measurements showed the ASBR was effective at increasing the pH overflows from the northeastern corner of the infiltration pond down a steep slope into Aspen Creek and reducing the oxidation state of Aspen Seep discharge. Burleson measured ASBR effluent pH between 6.5 and 7.0 during the 2011 treatment season. The pH of ASBR effluent appears to have remained relatively stable during 2011. ORP is another key indicator of bioreactor performance. Burleson measurements show a decreasing trend in ASBR effluent ORP from June through September. The lowest ORP measurement coincides with bioreactor flushing.

The following improvements were made to the ASBR during 2011:

- replacement of lead acid storage batteries with amalgamated glass mat storage batteries to reduce maintenance requirements
- construction of a separate battery room with a passive solar heating system in the electrical conex
- installation of an uninterruptible power supply (UPS) to improve power quality to ASBR treatment system control panel

Discharge of ASBR effluent from the infiltration pond to Aspen Creek continued during 2011. Discharge seeps through the downslope infiltration pond berm into Aspen Creek. Water also overflows from the northeastern corner of the infiltration pond down a steep slope into Aspen Creek.

Significantly above average precipitation was measured at Monitor Pass during the winter of 2010-2011. Discharge rates from the acid drainage sources on site during 2011 were similar to those observed during 2006, and historical flow rates, considering the amount of precipitation.

The range of pH measurements at the CUD and DS in 2011 was similar to the pH range observed during treatment seasons between 2001 and 2010. These pH trends may reflect flushing of oxidation products from the variably saturated volume of the subsurface materials during the high rainfall years (such as 2005 and 2006) and increased retention of material above the water table during drier years (such as 2007 and 2008).

Visual observations were recorded during 2011 water quality monitoring to supplement field screening data. Visual observations included creek water clarity, the presence of unusual materials within the creek, and the presence of aquatic life.

High flows that occurred throughout the late winter and spring during 2011 were cloudy and turbid in comparison to the clearer water of Mountaineer Creek that is not affected by Leviathan Mine.

The visual appearance of Leviathan Creek began to improve in late May 2011 after Atlantic Richfield began capturing CUD and DS discharge, and run off declined. Flow in Leviathan Creek downstream became clearer, and orange chemical staining in the creek bed began dissipating.

Seepage from the east bank of Leviathan Creek below and immediately up and downstream from the DS collection tank was observed since 2008 continued through the 2011 treatment season. Seepage flowed freely into Leviathan Creek for about 10 feet along the stream bank, and red staining was evident in the stream bank sediment. The presence of the seep suggests that shallow, acidic seepage is present that is not intercepted by the DS collection system.

Aquatic life in Leviathan and Bryant Creeks was documented during 2011 oversight visits. Trout was observed in the pool within Leviathan Creek at Station 17. Aquatic life observations were qualitative; thorough documentation of aquatic communities was outside the scope of oversight tasks. A separate ongoing aquatic macro-invertebrate study includes sampling in the Leviathan Mine watershed in the spring and late summer each year. The macro-invertebrate study quantifies the health of the aquatic community downstream from the mine.

Water treatment by the Regional Board and Atlantic Richfield in 2011 led to seasonal improvements in Leviathan Creek water quality downstream from the mine. Water quality monitoring and visual observations suggest that conditions in Leviathan Creek at the end of the 2011 treatment season were generally similar to conditions at the end of the 2010 treatment season. This summer was the eleventh consecutive season of CUD capture and treatment, and fifth consecutive season of DS partial capture and treatment.

Above average precipitation resulted in increased accumulations of water in on-site ponds compared to the accumulations observed the last couple of drier years, and resulted in increased discharge rates from acid sources during the 2010-2011 winter as compared to earlier years. The resulting increase in pond water levels did not restrict either the Regional Board or Atlantic Richfield from treatment system operations. Free-board capacity in Ponds 2, 2N, 2S and 3 was preserved, and overflow to Leviathan Creek prevented, by Spring Season treatment using an RCTS (Rotating Cylinder Treatment System) system. The storage capacity at Pond 4 was increased during the treatment season by removal of sludge.

The Regional Board treated all of the available water from Ponds 2N and 2S and Pond 1. The Pond Water Treatment System (PWTS) operated 24 hours per day, 7 days per week for the 43 day operating period in 2011. The switch to use of imported lime slurry in the PWTS resulted in increased truck traffic necessary to support treatment activity at Pond 1 during 2011 field activities.

Atlantic Richfield captured and treated CUD and DS discharges for 166 days in 2011, which led to noticeable improvements in Leviathan Creek. Atlantic Richfield initiated capture of the CUD and DS flows on May 4, 2011 prior to observation of significant decrease of water quality at Station 15. The HDS treatment system was commissioned, and operated into early November before

demobilization. At Aspen Seep, sludge was removed from the bioreactors, and dewatering via belt filter press was completed.

Following actions for the 2012 treatment season were recommended:

- The surface of mine waste (overburden and waste rock) north of the pit should be evaluated to identify the feasibility of modifying the surface to minimize infiltration and promote runoff of water. This activity would likely reduce the amount of acid drainage requiring treatment.
- Impacts of storm related runoff events should be assessed to evaluate their contributions to discharge of sediment, salts, and metals to the Leviathan Creek watershed.
- Potential for remobilization of the Leviathan Creek landslide as the beaver ponds expand and pond levels (and associated pore water pressure within the landslide toe increases) should be monitored.
- Sediment accumulated at the vicinity of the beaver ponds is not well characterized. Chemical precipitates may also be accumulating within the beaver ponds. Consideration the event of a beaver dam failure should be made.
- Improvements to the storm water controls at the site should continue as necessary to prevent future damage to treatment systems, reduce damage to roads, and minimize interference of run-off with treatment activities.
- CUD and DS capture should begin as soon as practicable in the LAS during 2012 to minimize the impacts of CUD and DS discharges on Leviathan Creek as spring flows decrease from spring runoff peak flows.
- Monitoring of water chemistry and metal content should be performed during base flow conditions at the Beaver Ponds to evaluate the chemical conditions and the potential for a source of acidity in the area.
- Traffic to and from the site should continue to be coordinated between the Regional Board and Atlantic Richfield contractors to minimize the chance for accidents and/or spills.
- The Delta Seep capture system should be evaluated and improved. Seepage of acid drainage into Leviathan Creek below the existing collection tank demonstrates that the existing collection system remains only partially effective.
- Improvement of the DS capture system should include evaluation of methods to direct runoff and sediment around the capture system. This would prevent loss of capture during runoff due to overwhelming the pump system capacity and clogging of the inlet pipes to the capture tank.



Figure B.8-4. Aspen Seep bioreactor settling pond: Note black sludge in middle-left. View to West. June 3, 2011.



Figure B.8-5. Aspen Seep bioreactor infiltration pond: View to North. June 3, 2011.



Figure B.8-6. Aspen Seep bioreactor sludge dewatering equipment: View to Northeast. July 13, 2011.



Figure B.8-7. Aspen Seep bioreactor settling pond prepared for sludge removal. August 18, 2011.



Figure B.8-8. Ponds at Leviathan Mine full of acid mine drainage during winter (with overflow pipes).

Acidic mine waste flows from at least four separate discharge locations. Sulfuric acid with dissolved metals and arsenic enter Leviathan and Aspen Creeks unless it can be captured and treated. Since the eastern slope is fairly arid, even the relatively low flow rates of the acid mine drainage have devastated the stream system as far away as the Carson River, nine miles downstream. Atlantic Richfield will conduct a feasibility study that will include an estimate of power needs for a long-term remedy. The Leviathan Mine project will require active remediation for a very long time. Because the site is remote and many miles from power lines, diesel or other fuel must be trucked into the site to power any cleanup activity. Snowfall and wet winter conditions limit large vehicle access on the steep mountain road to only about half the year during the dry summer months. Using renewable energy instead could potentially allow treatment of the acid drainage for longer periods of the year, would be a much greener cleanup option, and would reduce stress on local roads.

B.8.5 Regulatory Challenges

Most tribes have established Environmental Departments funded primarily by USEPA. As with states, USEPA can delegate programmatic authority to tribes for Environmental programs, including regulatory authority under the Clean Water Act, if the tribe applying for such authority has an established Environmental Department and satisfies other USEPA requirements. Consequently, many tribes have developed enforceable Water Quality Standards. Although the tribe has

developed its own Water Quality Standards, they are not yet enforceable until the tribe receives programmatic authority from USEPA. However, the tribe has been in close communication with those designing the Creek area BCR regarding the tribe's Water Quality Standards which reflect the tribe's priorities in addressing surface water contaminants (although the BCR may have the effect of treating water that meets Water Quality Standards, the tribe may not view the resulting water quality as being protective of the tribe, considering the tribe's more intensive use of water and water related resources).

B.8.6 Stakeholder Challenges

Most Native American tribes have a different perception of environmental protection and remediation than that of western European or modern American societies. Therefore, it is very important to approach the tribe(s) that may be affected by the proposed BCR, and initiate a consultation process before beginning even the conceptual design. Federally Recognized Tribes and USEPA have jointly developed consultation policies and procedures to be used before environmental regulations or actions, that may affect tribes, are initiated. Therefore, it is highly recommended that those proposing BCRs on, or near, tribal lands or in areas of tribal jurisdiction, contact the appropriate USEPA Regional office and ask for office of the USEPA tribal liaison official who will likely offer detailed guidance on the appropriate tribal consultation procedures.

B.8.7 Other Challenges and Lessons Learned

A bioreactor treatment system, ranging from pilot-scale to full-scale implementation, has been operating at the Leviathan Mine since 1993. The technology has experienced many changes since its inception. In its current form, this treatment technology operates year-round and is compliant with USEPA discharge standards for all target metals except iron.

As the site remedial project manager explains, initially designed with simplicity in mind, the Aspen Seep SRB (Sulfate Reducing Bioreactors) has required more operator involvement than originally anticipated because pumping is required to keep the system operating properly. Although the system has proven to be both effective and reliable, it has also required more maintenance than originally planned.

The remote nature of the site and the surrounding environment are impacting system operations. The climate is also influencing the system with slower biological activity during the winter months. Winter snowpack limits access to the site for eight months out of the year requiring operating materials such as sodium hydroxide, ethanol, and diesel fuel to be stored in bulk before the winter. Similarly, equipment replacement, sludge dewatering, and sludge transfer are all performed during the summer months.

This technology can now be implemented at other sites. The bioreactor system at Leviathan Mine successfully addressed problems related to carbon availability and sulfate reduction. However, due to the unique characteristics of each site, the dose for base and ethanol would need to be determined through a simple bench test

The capitol costs for construction of the gravity-flow operation amounted to $836,600 and changing to the recirculation mode added nearly $30,000, for a total of $864,100. Operating at an average flow rate of 10 gpm, the operation and maintenance costs of the system are $15.73 per 1,000 gallons of treated ARD. The compost-free sulfate-reducing bioreactor at the Aspen Seep is the first of its kind. The project was labor intensive because the operating conditions at the site were continually altered to stress and test the system. The system also included many optional features, such as controlling and routing flows, that would not normally be used in most systems. Because of these factors, the final costs are likely to be higher at Leviathan than the cost of operating and maintaining the system at other sites.

The capital cost of the rotating cylinder treatment system (RCTS) was less than $100,000. The cost of operation and maintenance depends on the volume of water treated each season. Overall treatment cost of all technologies applied at the Leviathan site is reported as follows:

- Capital: $5 million
- Operation and maintenance: $2 million annually

Unpredictable amounts of acid mine drainage from year to year make budgeting and contracting processes very uncertain.

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B.9 Wheal Jane Mine Case Study

Pilot Passive Treatment Plant, Cornwall, United Kingdom

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B.9.1 Site Information

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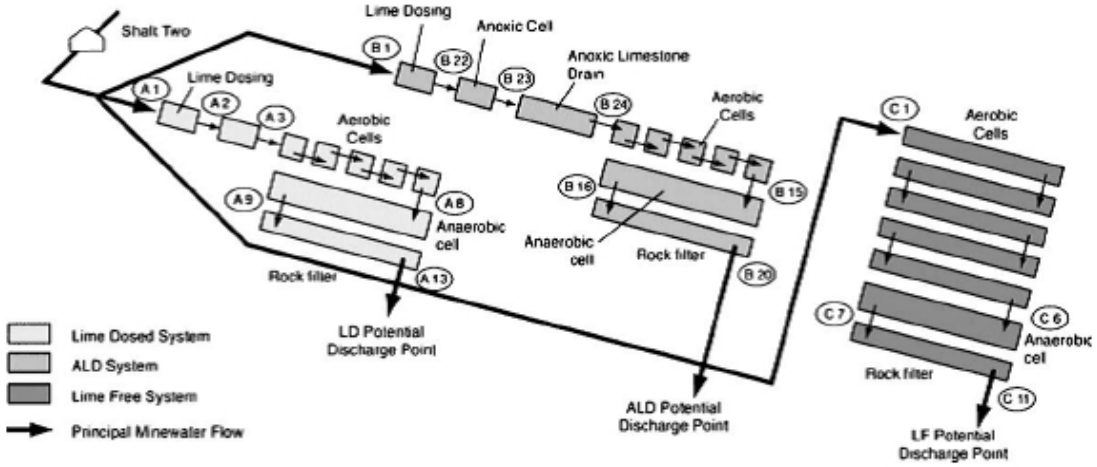
Name, Location, and Site Description

The Wheal Jane tin mine is located within the Carnon River valley in Cornwall, England, approximately seven miles southwest of the town of Truro. Wheal Jane was one of the last tin mines to be operated in Cornwall, with mining on the site first recorded in 1740. Underground operations ceased in 1991 as a result of the fall in tin price when it was closed and abandoned under the Mines and Quarries Act.

Closure of the mine resulted in the termination of mine dewatering operations and a rise in water levels. In January 1992, there was a sudden and uncontrolled release of highly contaminated mine water into the Carnon River and Fal estuary. The contaminated water was extremely acidic, and contained high concentrations of iron, zinc, cadmium, and arsenic. Oxidation of the iron rich mine water generated orange-brown discoloration over a downstream drainage area of more than 6.5 x

106 m² including parts of Falmouth Docks. The highly conspicuous nature of the contaminated plume led to the event attracting worldwide media attention (Brown et al. 2002).

The potential use of passive treatment technology for the treatment of mine water was assessed through the operation of three pilot treatment systems since 1994 (see Figure 1). Data was collected on system performance after start-up which indicated poor performance. The passive treatment system was renovated and studied intensively by researchers from several universities from 2000 to 2002. An active lime treatment plant capable of treating all discharge began operation in 2000. Studies showed there is insufficient land area available to contain a passive treatment system. The passive system is no longer operational due to lack of funding.



B.9-1. Wheal Jane pilot systems.

Source: Reprinted from Whitehead and Prior, 2005.

B.9.2 MIW Chemistry

Water from the Wheal Mine is extremely acidic and contaminated with high concentrations of iron, zinc cadmium and arsenic (Table B.9-1).

Table B.9-1. Wheal Jane water chemistry 2000-2002 (Doshi 2006)

Analyte	Influent (mg/l)
pH	3
Al	12.4
As	9
Cu	0.1
Fe	161
Mn	5.3
Pb	0.1
Zn	41.9
Sulfate	1094

B.9.3 System Design

B.9.3.1 BCR Design

Three parallel anaerobic BCRs were constructed to receive flow from a series of aerobic cells) that were designed based on USBM criteria to remove iron at low pH (see Figure 1). The maximum design flow capacity is approximately 9 GPM and can receive short term flows of up to 30 GPM. This represents less than 1% of the total mine water discharge from the Wheal Jane mine. The BCRs were operated in a downflow configuration.

The plant contained three multi-cell treatment systems that used one of three pretreatment methods to raise pH: lime dosing to pH 5.0 with calcium carbonate (LD), an anoxic limestone drain (ALD), or a lime-free system without pretreatment (LF). Due to plugging, the ALD system was modified to function as another lime dosing system. In each treatment system, the pre-treated drainage passed to aerobic reed bed wetlands for iron and arsenic removal. Next, water flowed through an anaerobic cell for sulfate reduction and heavy metals removal. The final stage was an aerobic rock filter, designed to promote manganese removal. (Whitehead and Prior 2005).

The anaerobic cell in each system was approximately 87.5 m long, 8.75 m wide, and 1 m deep, lined with high-density polyethylene (HDPE) membranes. This depression contained a mixture of 95% softwood sawdust, 5% hay, and a small quantity of cow manure to inoculate the bioreactor with SRB. The mixture was covered with approximately 0.4 m of earth and gravel, to maintain anaerobic conditions. (Johnson and Hallberg 2005). Pilot cells began operation in 1995, and the compost in the bioreactors was replaced in early 2000. The compost was amended with limestone and re-inoculated with manure (Guzek 2012).

B.9.4 BCR Performance

The bioreactors were constructed primarily to generate alkalinity, raise pH, and remove heavy metals, particularly zinc. The influent acid mine drainage (AMD) from the aerobic wetlands was similar for each system, with a pH between 3 and 4, and 95 to 100% dissolved oxygen saturation. Redox potential and dissolved oxygen concentrations decreased in all three bioreactors (Johnson and Hallberg 2005).

All three bioreactors increased pH, to 5.5 (LD system), 5.0 (ALD system), and 5.9 (LF system). When discharged to the rock filters, pH decreased to 4.5 in the LD and 5.0 in the ALD systems, while it increased to 6.8 in the LF system (Table B.9-2). Oxidation of excess sulfide and ferrous iron from the bioreactor effluent may have contributed to the downstream pH decrease (Johnson and Hallberg 2005). The low pH in the bioreactor effluent and rock filters likely contributed to the failure of manganese removal in the LD and ALD systems (Whitehead et al. 2005).

Table B.9-2. Wheal Jane bioreactor water chemistry, 2001-2002, all concentrations in mg/l (Johnson and Hallberg 2005)

	LD		ALD		LF	
Analyte	Influent	Effluent	Influent	Effluent	Influent	Effluent
pH	3.6	5.5	3.6	5	3.9	5.9
Fe	3.8	17.3	5.8	12.6	3.6	0.4
Zn	36.4	16.3	29.7	9.8	40.7	0.01
Sulfate	245	205	230	184	233	128
ORP (mv)	768	178	743	222	741	66

Sulfate concentrations decreased in all three bioreactors over the treatment period. During the entire monitoring period, the bioreactors lowered sulfate concentrations by 27±18% (LD), 23±12% (ALD), and 62±20% (LF). The bioreactor in the LF system consistently removed the most sulfate. However, its efficiency decreased from 91% to 39% over the study period. In all systems, sulfate concentrations increased from bioreactor effluent concentrations in the rock filter. (Johnson and Hallberg 2005)

Like sulfate, zinc was removed in all bioreactors, though the LF system bioreactor had the greatest removal rate (Table 2). Based on September 2002 measurements, the bioreactors removed 55% (LD), 67% (ALD), and 99% (LF) of soluble zinc. Dissolved zinc concentrations increased in the rock filter in the ALD system, but decreased in the other systems (Johnson and Hallberg 2005). However, data collected from March 2000 to February 2002 showed removal rates over 99% for the LD system (Table 3; Whitehead 2006).

Though much soluble iron was removed in the pretreatment and aerobic cells, concentrations continued to change in the anaerobic bioreactors. Soluble iron in the LD and ALD bioreactor effluent was more than twice the influent (Table 2). Soluble iron decreased within the LF system bioreactor and generally remained low. All three complete treatment systems ultimately lowered soluble iron concentrations by at least an order of magnitude. (Johnson and Hallberg 2005)

Though excess sulfide was produced in all three bioreactors, it did not effectively form iron sulfide precipitates in two bioreactors, apparently due to the low pH. Furthermore, ferric iron minerals entered the bioreactors and were reduced to soluble ferrous iron, increasing dissolved iron within two of the bioreactors. Sulfides and ferrous iron exported in the bioreactor effluent were then oxidized to sulfate and ferric iron, increasing acidity downstream. (Johnson and Hallberg 2005)

Pilot cells began operation in 1995, and the compost in the bioreactors was replaced in early 2000 to improve performance in raising pH and removing metals. A number of conditions combined to overwhelm the BCR cell receiving the effluent from the aerobic cells. First, the aerobic cells were not as efficient as expected in neutralizing mineral acidity, and rainfall dilution did not significantly affect the mineral acidity of the water, a critical design parameter for BCR cells. Second, overloading occurred during the winter when bacterial activity was apparently stressed already due to the low water and air temperatures. Third, and lastly, the initial organic substrate did not contain any inherent buffering capacity (bench-scale tests had not been performed due to schedule restrictions). In summary, the stressed organisms were hit with an acidity overload and there was no self-buffering component in the substrate to counter it. Consequently, the metal removal performance of the cell suffered. Fortunately, this was a pilot test, and the situation was corrected by excavating the anaerobic substrate, amending it with limestone, re-inoculating with manure, and installing a flow restriction device (orifice) on the aerobic cell outfall that helped to manage the flow peaks. The cell responded favorably and was subsequently more successful at zinc (and iron) removal.

Elimination of flow into the lime free system improved performance of the anaerobic cell. The LF system was shut down due to a fracture in the main influent pipe between 2000 and June 2001. During this time, flow was shut off to the lime free system. Upon recommencement of mine water flow, it was noticed that the effluent from the anaerobic cell had a different chemistry from the effluent of the other two cells which continued to receive flow. The pH of the discharge from the lime free cell was in the range of pH 6-7 as opposed to pH 4-5 for the other two anaerobic cells. Metal removal efficiency was also higher in the lime free cell. The shutdown possibly eliminated stresses to the population caused by low pH water which allowed the development of a more robust microbiological population that produced alkalinity, raising the pH and more efficiently removing metals.

B.9.5 Challenges and Lessons Learned

Johnson and Hallberg (2005) attribute the poor performance of the anaerobic cells to the following factors:

- Acidic, oxygenated water entering the bioreactors created suboptimal conditions for SRB growth.

- SRB populations were unable to tolerate the acidity of the system. A ten-month shutdown of the LF system may have allowed more tolerant microbial populations to develop and flourish in the anaerobic cell.
- Refurbishing the substrate with limestone increased the pH which was beneficial to the micro-organism community (Guzek 2012).
- Soluble and particulate ferric iron entering the bioreactors exceeded the capacity of the system to reduce and precipitate iron, particularly under low-pH conditions.
- Collection of rainwater in aerobic cells added unnecessarily to the flow entering the bioreactors.

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B.10 Peerless Jenny King Site Case Study

Upper Tenmile Creek, Helena MT

Acknowledgements

The team would like to acknowledge David Reisman, David Shanight, and Angela Frandsen from CDM Smith who submitted the Peerless Jenny King (PJK) Biochemical Reactors Case Study.

B.10.1 Site Information

Contacts

The USEPA remedial project manager for the Upper Tenmile Creek Mining Area Site is Tillman McAdams. The site contractor is CDM Smith, and the project manager is David Shanight.

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Name, Location, and Site Description

The Upper Tenmile Creek Mining Area site is located in the Rimini Mining District, southwest of Helena, Montana. It consists of numerous abandoned and inactive hard-rock mine sites that produced gold, lead, zinc and copper. Mining began in the district before 1870 and continued through the 1920s. Little mining has been performed there since the early 1930s. USEPA added the Upper Tenmile Creek Mining Area Site to the Superfund National Priorities List on October 22, 1999, due to mining waste problems in the 53-square-mile watershed. The small historic mining community of Rimini is located within the Superfund site boundaries. The site includes the drainage basin of Tenmile Creek upstream of the Helena water treatment plant and includes tributaries that supply water to the plant's five intake pipelines. USEPA identified 150 individual mine sites within the watershed boundary, of which 70 have been prioritized for cleanup. Many of these mine features are above the five City of Helena drinking water intakes, which supply about 50 percent of the city's water.

The Peerless Jenny King (PJK) project site is named after three mines in the area of the discharging adit. Most likely this draining adit was connected to the original Peerless Mine. The adit is the major flow in this area, but smaller seeps can be seen coming from both hillsides.

B.10.2 Chemistry

The PJK biochemical reactor (BCR) was designed and built by Jim Gusek of Sovereign Consulting Inc., David Reisman, USEPA Office of Research and Development, and a small contractor

crew from the on-site contractor, Envirocon. The entire system was designed and built in less than 30 days at the end of the field season in 2003.

The adit discharge chemistry varies throughout the treatment system. Water flows from the adit continuously, but the flows increase during the spring as snow melt percolates into the mine workings and the adit flow increases to above 70 gpm. During the summer season the flow is usually in the 5-10 gpm range, but can increase with storm events.

In general, PJK adit water quality has a pH ranging from 5.7 to 6.8, an average specific conductance of 324 µS/cm, and is consistently aerobic. The adit water is dilute acid drainage with relatively low levels of metals. On average, the water contains zinc (up to several milligrams per liter), iron (0.42 mg/L), aluminum (0.062 mg/L), copper (0.036 mg/L), nickel (0.0066 mg/L), and cadmium (0.0074 mg/L). Zinc is the primary metal of concern at the site.

B.10.3 System Design

The treatment layout was designed to take advantage of natural treatment processes already occurring at the site, and to allow for adequate metal removal treatment in the limited area available. The treatment layout, shown on Figure B.10-1, consists of an aerobic iron depositional wetland, a cascading connector channel with some limestone, and four BCR cells in series, followed by emptying of the BCR effluent either by draining from the fourth cell or percolation into the groundwater. All of the effluent eventually enters a lower natural wetland area where additional deposition and precipitation occurs.

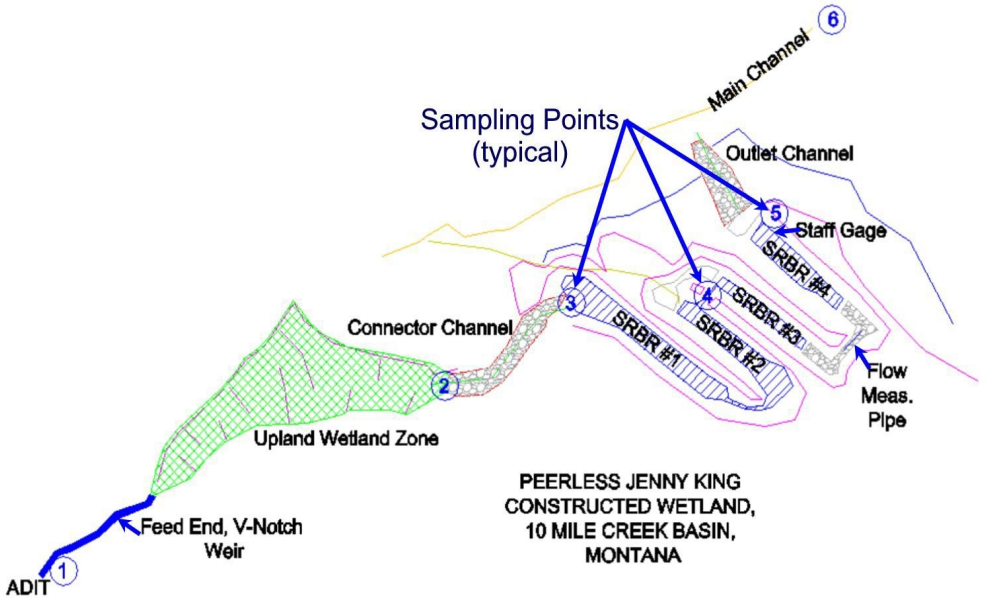


Figure B.10-1. Schematic of the PJK bioreactor.

Mine drainage exits the adit and flows through a wetland in order to provide pretreatment and precipitate iron oxides. Hay bales are used as baffles in the wetland to help slow the flow rate. Several metal removal processes occur in aerobic wetlands including:

- Precipitation as insoluble metal sulfides, carbonates, and hydroxides;
- Cation exchange and chelation with wetland soils;
- Sorption of metals to organic matter and iron hydroxides;
- Metal removal via plant uptake.

Pre-treated water then passes through a channel before entering four compost-based serpentine sulfate-reducing bioreactors in series. The BCR cells use bacteria to treat the semi-acidic metal-laden water. The cells are filled with an organic media mixture, which includes shredded wood and manure. The depth of the compost varies from 2 ft to 3 ft in the four reactors. Dairy manure was used as the bacterial inoculum. The mine influence water (MIW) pH was determined to be high enough to preclude the need for limestone to protect the sulfate reducing bacteria from excess acidity. Sulfates are typically about 100 mg/L and the pH is approximately 6.0 at the influent of the first bioreactor.

B.10.4 BCR Monitoring and performance

B.10.4.1 Sampling Locations

There are six surface water sampling locations throughout the treatment system as shown in Figure 1. Beginning at the adit and proceeding through the system, the sampling locations are listed below.

Sampling locations

- Station 1 – Adit discharge.
- Station 2 – Upper wetland. This sample is taken at the outlet of the wetland prior to the cascading channel.
- Station 3 – listed as SRBR 1. This sample is taken at the inlet of the first SRBR cell.
- Station 4 – listed as SRBR 3 inlet. This sample is taken at the inlet of SRBR 3.
- Station 5 – listed as SRBR 4. This sample is collected at the outlet of SRBR 4 if there is flow detected.
- Station 6 – Main channel. This sample is collected in the drainage that receives discharge water from the treatment system. The sample location is approximately 25 yards downstream from the outlet of the treatment system. The treatment system discharge is a small fraction of the flow at this sampling point. Discharge from the treatment system typically infiltrates into the soil surrounding cell 4.

In addition, shallow well points (2-foot pipes) were included to allow monitoring of the subsurface zone of the wetland.

B.10.4.2 Data summary

Sampling and analysis of the PJK water has been done from 2003-2010, and some winter sampling has been done. Sampling at all the locations above was done bi-weekly for 2005-2006 after the BCR had been functioning for several years. A summary of the data follows.

Data summary by analyte

- pH − The Station 1 pH ranged from 5.7 to 6.8. In general, pH values increased slightly in the treatment system; the Station 4 pH ranged from 6.2 to 7.5. The pH increase is likely due to the buffering of the influent water by alkalinity generated by sulfate reducing bacteria contained in the SRBR cells, since there is no limestone for alkalinity in the cells.
- Oxidation-Reduction Potential (ORP) – Subsurface ORP measurements were recorded throughout the 2006 sampling season and were consistently negative. The negative ORP conditions in the well points associated with SRBR cells 2 and 3 are indicative of anaerobic conditions favorable for bacteria-mediated sulfate reduction. The positive ORP values observed in MW-SRBR4 are consistent with the low water level in that cell as Station 5 was dry for much of the summer. During the winter sampling event (January 24, 2006), subsurface ORP values were positive. This suggests that bacterial activity decreases in the winter months, probably in response to depressed ambient and water temperatures. By comparison, ORP values measured at the surface sampling sites during the summer season (Stations 1 - 6) were typically positive. This is expected as surface water is continuously exposed to atmospheric oxygen.
- Temperature– The Station 1 water temperature ranged from 41 to 45°F (5.0 to 7.2°C). The minimum water temperature was measured on June 1, 2006.
- Specific conductance – The average Station 1 and Station 4 specific conductance values were 323 and 257 micro-Siemens per centimeter (µS/cm), respectively. This decrease in specific conductance is consistent with the decrease in metals and sulfate concentrations.
- Field Sulfate Concentrations and Trends – Sulfate was measured in the field with a Hach kit, which has been reported to be conservatively low compared to analytical methods (Reisman et al, 2007). For comparison purposes, it is sufficient for discussion. Station 1 and Station 4 sulfate concentrations averaged 78 mg/L and 65 mg/L, respectively, during 2005. The main BCR cells' sulfate values displayed a generally decreasing trend from June 27 through October 5, 2006. The observed MIW sulfate values were consistently less than the surface water values. This is consistent with the ORP values discussed previously and indicates that sulfate reduction is preferentially occurring deeper within the BCR cell media.

Table B.10-1. Data Summary

Analyte mg/l	Influent Station 1	Leaving UWZ Station 2	Influent BCR Station 3	Mid-BCR Station 4	End BCR Station 5
pH	6.75	7.07	7.17	7.15	7.21
Ca	34	36	35	36	32

Analyte mg/l	Influent Station 1	Leaving UWZ Station 2	Influent BCR Station 3	Mid-BCR Station 4	End BCR Station 5
Cd	0.011	0.011	0.0045	0.0030	0.0061
Fe	0.33	0.4	0.0	0.1	0.1
Mn	4.6	4.7	1.9	1.5	0.3
S	31	31	30	31	26
Sulfate	94	92	91	93	78
Zinc	1.35	1.35	0.62	0.31	0.87

B.10.4.3 Microbiological Study at Upper Tenmile

The BCR cells are unique horizontal flow reactors with high-permeability zones at the entrance and exit of each cell to maintain flow. The microbial communities of two field-scale pilot sulfate-reducing bioreactors treating MIW, PJK, and the nearby Luttrell BCR were compared using bio-molecular tools and multivariate statistical analyses (Hiibel et al. 2008). The two bioreactor communities were found to be functionally similar, including cellulose degraders, fermenters and sulfate-reducing bacteria (SRB). Very few of the existing human constructed BCRs are horizontal flow, yet many natural wetlands are shallow, horizontal flow systems (including the natural wetlands at the end of the PJK system).

The microbial study served multiple purposes. Little research had been done on the microbial consortium inhabiting these previously misnamed sulfate-reducing bioreactors. The bacteria in these reactors are of over 90 percent other types of bacteria, and there are many different functional groups that play a role in BCR actions and survival. Second, most human-constructed BCRs are almost completely anaerobic to promote the chemical process leading to precipitation. This pilot study was to test the metal removing efficiency of the BCR cell/system where the top of the cell is constantly aerobic. Finally, the answer to many of these questions was important to USEPA because leaking adits in this and other mine-impacted areas have very low flow, and USEPA needed to develop passive systems that could remove metals with low cost, low maintenance biological treatment systems. Many of these sites are not suitable for vertical reactors, and USEPA ORD wanted to study the ability of horizontal, small area treatment systems to remove metals from smaller flow MIW areas.

Luttrell and PJK were ideal field-scale systems for the USEPA ORD comparative microbial study because of their similar geographic location and climate. Furthermore, both systems were constructed with the same organic material and were inoculated with manure from the same source, and thus the starting microbial communities at each site were highly similar. The differences between the two sites include the quantity and quality of influent MIW, the configuration of the bioreactors (horizontal flow at PJK versus vertical top-down flow at Luttrell), and the degree of exposure to aerobic conditions. The results were very informative. As a result of seasonal flow fluctuations and its design, several locations within the PJK bioreactor are known to have experienced

periodic aerobic conditions (especially the fourth cell which is only seasonally filled). It was hypothesized that this exposure to oxygen would result in differences in the PJK microbial community compared with that of the more consistently anaerobic Luttrell bioreactor, namely the presence of aerobic, facultative aerobic or aerotolerant species. The results of this study indicate correlations between microbial populations and remediation performance, suggesting that microbial communities play an important role in determining the success of the treatment process.

Over their lifetimes, the Luttrell bioreactor has removed more than 60 percent of the influent sulfate compared with the less than 40 percent removed at PJK. Neculita and colleagues (2007) documented a wide range of sulfate removal rates for passive sulfate reducing systems treating MIW and operating under a variety of conditions, configurations and scales. These wide varieties of designs with no apparent trends with regard to remediation performance also suggest that the bioreactor's microbial community has a strong impact on the success of the treatment process. What was never studied due to funding limitations was the effect of introducing white rot fungi (WRF) to the aerobic portion of the PJK BCR cells. WRF are aerobic fungi, which are known to affect the cellulose-lignin bonds, and assist in the degradation of woody material, thus freeing up additional carbon for the SRB and other microbes in the cells. Although oxygen exposure at PJK is clearly an undesirable attribute, it is important to note that the bioreactor is still removing large quantities of sulfate and metals. The design and configuration of PJK also offer some advantages over Luttrell. Whereas flow at PJK goes freely through a wetland and alkaline channel, the Luttrell bioreactor is more closely controlled because flow leaves the repository and remains in piping, with the final section (~100 yards) in narrow 1-inch piping to help regulate water entering the bioreactor. In a stark comparison, and what has been enlightening to see at PJK, is the natural expansion of sulfate-reducing areas of the upper wetlands surrounding the straw bales, which were originally used to delineate the wetlands. This natural expansion is an extra benefit of this type of treatment system, and is perfect for areas with low flow adits and seeps, which was one of the objectives of this treatment system and the USEPA ORD study. Therefore, a design similar to PJK may still be an attractive approach for the remediation of large volumes of less-severe MIW. In general, the PJK bioreactor treats a larger volumetric flow rate while the Luttrell bioreactor treats a more acidic and heavy metal-laden MIW. The conclusion derived from the PJK pilot demonstration and the ORD studies is that passive treatment BCRs can be designed to meet site-specific needs, and can remove metals from MIW even in remote locations difficult to access and without power.

B.10.5 Regulatory Challenges

None.

B.10.6 Stakeholder Challenges

None.

B.10.7 References

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B.11 Golinsky Mine Case Study

Shasta County, California

Acknowledgements

The mining team would like to acknowledge Christine Brown who submitted the Biochemical Reactors Case Study. The information in this case study is for the Biochemical Reactors used to Treat Mine Influenced Water prepared by the ITRC Mining Team 2013.

B.11.1 Site Information

Contacts

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Name, Location, and Site Description

The Golinsky Copper Mine operated between the 1890's and the 1930's, where sulfide ores of copper, zinc and minor amounts of precious metals were mined and smelted on site. The site is located approximately 2 miles from Little Backbone Bay in a remote area near Shasta Lake, California (see Figure B.11-1). Three mine adit portals (Upper Portal, Lower Portal, and Portal 3) discharge mining-influenced water to Little Backbone Creek, which is a tributary to Shasta Lake. The site is at the edge of the Klamath Mountains U.S. Forest Service property and is approximately 20 acres in area. Access is limited to boat and All-Terrain Vehicle (ATV).

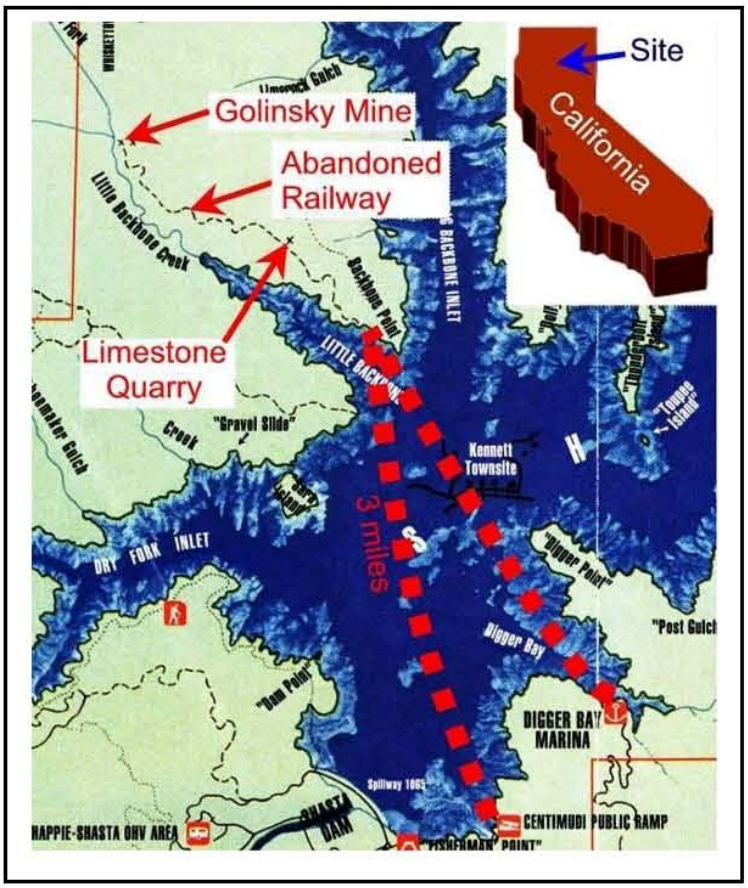


Figure B.11-1. Site location map.

Source: Gusek et al. 2011.

B.11.2 MIW Chemistry

Chemical analyses of the mine discharges revealed that the main mine pool had pH ranging from 2.5 to 4 and contains heavy metals including iron, aluminum, copper, cadmium and manganese (Gusek et al. 2011).

B.11.3 System Design

B.11.3.1 BCR Design

The full system was planned to consist of several BCR cells needed to treat the total flow from the Golinsky mine, and the construction of the system would be implemented in phases to allow for further collection and analysis of data, and challenges to mobilizing a large construction project in a remote location with limited access. The first phase of the system, the module 1 BCR, was constructed in 2010. The system schematic is presented in Figure B-11.2.

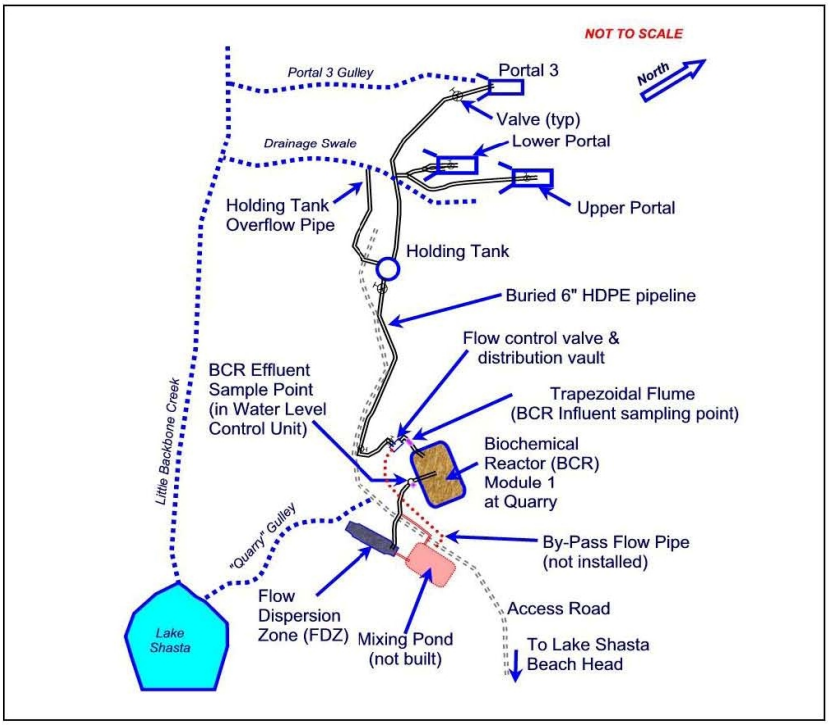


Figure B.11-2. Golinsky BCR System Schematic

Source: Gusek et al. 2011.

One of the unique features of the Golinsky BCR is that the since there was no space to construct the BCR near the mine portals, the BCR was constructed in a limestone quarry approximately 1.5 mile away. A pipeline system was thus constructed to collect the mine discharge and pipe it to the BCR system.

The module 1 BCR cell is designed to be a vertical flow reactor with flow entering the top of the cell and flowing out the bottom. Flow into the BCR is controlled via a flow distribution vault and trapezoidal flume. The design flow rate is approximately 38 L/min. A design metals removal rate of 0.3 moles/day/m^3 was used. The substrate consists of rice hulls (10%), wood chips (49.7%), hay (10%), limestone (30%), and manure (0.3%). A total of 1,264 m^3 (1,652 yd^3) of substrate was added to the BCR. Included in the 0.3% manure was about 1.53 m^3 of substrate saved from decommissioning of the pilot-scale reactor, with the belief that microbes present in the pilot substrate, having already adapted to the site conditions, would decrease the time for the BCR to reach maturity.

The module 1 BCR is enclosed by earthen berms within the limestone quarry (see Figure B.11-3).

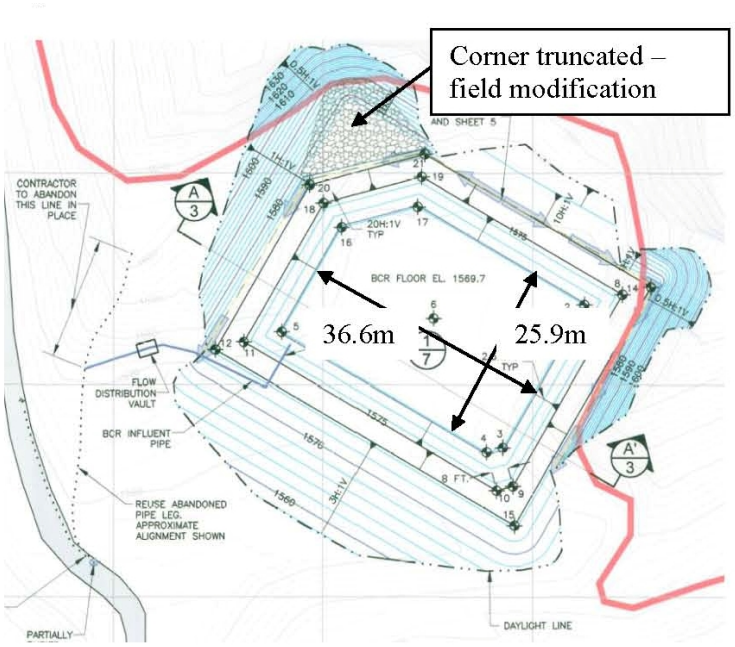


Figure B.11-3. BCR earthwork in limestone quarry.

Source: Gusek et al. 2011.

The organic substrate layer was designed to be 0.9 m thick, underlain by a 150 mm (6-in) gravel drainage layer containing a network of perforated pipes. The pipe network subdivided the floor of the BCR into four equal zones to minimize short circuiting. The final design would include 0.152 m (6-in) of standing water above the substrate and 0.5 m (1.64 ft.) of freeboard. The earthwork was lined with a geomembrane comprised of linear low density polyethylene (LLPDE), 60 mil (1.5 mm) thick. The as-constructed total floor footprint was approximately 3.6% less than design due to local geotechnical conditions. The reduction in theoretical flow capacity of 1.4 L/min falls within the level of confidence in the original design.

B.11.3.2 Pre- and posttreatment requirements

No specific pretreatment processes are needed; the mining-influenced water (MIW) is collected from the three portals by a collection system, which transfers the MIW via gravity to a holding tank. The MIW is conveyed to the BCR approximately 1.5 miles away by a mine water delivery pipeline.

Post treatment consists of discharging the effluent from the BCR into local soils in a flow dispersion zone (FDZ) as shown in Figure 2. The FDZ includes buried infiltration chambers, piping, and drainage gravel. The FDZ further consists of six 20-ft long, 8-ft wide benches that are sloped at approximately 1V:10H and are separated by 2-ft drops at 1H:1V slope. The FDZ follows the approximate hillside contour to the west. The top bench includes a deep zone comprised of three layers of buried infiltration chambers surrounded by riprap.

The pH neutral, metal-free, infiltrated water would travel through the subsurface in bedrock fractures or along the bedrock/colluvial surface before entering Lake Shasta as a nonpoint source.

B.11.4 BCR Performance

The design objectives of the Module 1 BCR are to:

1. Remove heavy metals (such as iron, copper, zinc, cadmium) as sulfide precipitates.
2. Remove aluminum as a hydroxy-sulfate.
3. Remove sulfate by reduction to hydrogen sulfide.
4. Maintain the pH at a value of 6 or above by adding alkalinity to the water in the form of bicarbonate.

Preliminary performance data from March 2011 indicated that the BCR removed about 91% of the influent-combined metals load and pH increased from about 2.8 to 6.6. The sulfate reduction rate was 0.09 moles/day/m^3, which was greater than the metals loading of 0.07 moles/day/m^3. These preliminary performance values agree with the pilot BCR data.

B.11.5 BCR Monitoring

Monitoring is conducted at three locations, the BCR influent at the flume or flow distribution vault, the BCR effluent at the Water Level Control Unit, and at the FDZ prior to infiltration into the lowermost portion. The BCR influent is monitored for alkalinity, pH, ORP, conductivity and temperature (field measurements), and laboratory analyses for sulfate, Ca, Mg, and dissolved metals (Al, Fe, Cu, Zn, Cd, Mn). The purpose of monitoring these parameters in the BCR influent is to establish baseline conditions.

The BCR effluent is monitored for these same parameters as well as biological oxygen demand (laboratory measurement). The purpose of measuring Ca and Mg in the effluent is to monitor limestone consumption. The purpose of measuring sulfate and dissolved metals in the effluent is to evaluate metal removal efficiency. The purpose of measuring biological oxygen demand (BOD), alkalinity, pH, ORP conductivity and temperature in the effluent is to evaluate BCR performance.

The FDZ is monitored for the same parameters as the BCR influent, with the purpose of evaluating FDZ performance.

B.11.6 Regulatory Challenges

None reported.

B.11.7 Stakeholder Challenges

None reported.

B.11.8 Other Challenges and Lessons Learned

There were a number of construction challenges. The remote location of the site made selection of a suitable lake-side mobilization site difficult. The solution was to use a United States Forest Service boat ramp, located 4.5 miles away, which added to travel times and prolonged the project schedule.

The construction of the mixing pond was deferred due to construction delays associated with the project. A temporary 25 mm (1-in) reinforced hose was installed to convey BCR effluent from the BCR outfall pipe to the flow dispersion zone. This installation plugged quickly due to apparent damage from a curious bear, as evidenced by telltale bite marks. The hose was subsequently replaced with solid 76 mm (3-in) PVC pipe.

After the BCR was constructed, but before the mining-influenced water could be introduced, seasonal rainfall filled the BCR. To address the problem, flow of Portal 3 water, which had very low concentrations of sulfate (8 mg/L) and no acidity, was introduced. To encourage sulfate-reducing bacteria growth, about 9.1 kg (20 lb.) of Epsom salts were added to the BCR in a single dose and a 13.6 kg (30 lb.) tea bag of gypsum was suspended in the BCR as well. In January 2011, the flow from Portal 3 was disconnected and the flow from the Lower Portal was connected, and the tea bag of agricultural gypsum was removed.

B.11.9 References

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B.12 Dänkritz 1 Uranium Remediation Case Study

Sachsen, Germany

Acknowledgments

The mining team would like to acknowledge H. Eric Nuttall, ITRC member who submitted the Biochemical Reactors Case Study. The information in this case study is for the Biochemical Reactors used to Treat Mine Influenced Water prepared by the ITRC Mining Team 2013.

B.12.1 Site Information

During the Cold War, uranium mining, milling, and related processing activities were conducted with little care for the environment. Large quantities of liquid and solid radioactive and hazardous wastes were deposited in unlined surface sites. In many cases, these wastes have contaminated the soil and the ground-water, and contaminants have migrated away from the site.

This pilot study focuses on the long-term stability of uraninite (UO_2) in the underground at a large tailings pond, 'Dänkritz 1' in Germany. The research studies for this pilot plant investigation were initially conducted at the University of New Mexico in Albuquerque, New Mexico and then followed up with large bench scale multiple year experiments in Germany.

Hence there are numerous potential sites in this mining region of Germany where BCR technology could potentially be successfully applied.

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Name, Location, and Site Description

Dänkritz 1, Sachsen, Germany

This pilot study focuses on the long-term stability of uraninite (UO_2) in the underground at a large tailings pond.

B.12.2 MIW Chemistry

Many radionuclides and metals ions can be converted from soluble to minerals by changing the valence of the metal ion. In the case of uranium the higher +6 valence is soluble in groundwater and migrates to form contaminated plumes and problems with drinking water. The current USEPA MCL for uranium in drinking water is 30 ppb. Many laboratory studies have shown that soluble uranium (+6) can be biologically transformed to insoluble uranium (+4), the mineral uraninite.

The chemistry used specifically in this case study to remove uranium by mineral precipitation is to anaerobically treat the groundwater using an appropriate carbon substrate. Substrates can be either liquid or solid carbon sources. The key to this specific approach is to biologically reduce the uranium to U^{+4} (the mineral uraninite) and then keep it in a reduced state by also forming iron sulfide which functions as a very effective oxygen getter thereby preventing the uraninite from reoxidizing to soluble U^{+6}. Key reactions are shown below.

Oxyen and nitrate removal—if present:

$$2O^{=} + NO_3^- \xrightarrow{Denitrifying\,bacteria} O_2 + N_2 \uparrow$$

Reduction of Metals and Sulfate:

$$SO_4^{2-} \xrightarrow{SRB} HS^-$$
$$M^{2+} + HS^- \rightarrow MS \downarrow + H^+$$

Uranium Reduction to insoluble uraninite mineral:

$$UO_2^{2+} \xrightarrow{SRB\,or\,FRB} UO_2 \downarrow$$

B.12.3 BCR Design

B.12.3.1 Bench Scale Test

The testing consists of long-term bench scale experiment followed by a pilot scale field investigation. A single-pass flow-through experiment was run in a 100-liter column: bioremediation for 1

year followed by infiltration of tap water (2.5 years) saturated with oxygen, sufficient to oxidize the precipitated uraninite in two months. Instead, only 1 wt.% uraninite was released over 2.4 years at concentrations typically less than 20 µg/L. Uraninite was protected against oxidation by the mineral mackinawite (FeS), a considerable amount of which had formed, together with uraninite. Figure B.12-1 is an Eh-pH diagram illustrating that iron sulfide precipitates at a lower Eh than does uraninite, preventing the uraninite from re-oxidizing the uranium from U^{+4} to soluble U^{+6}.

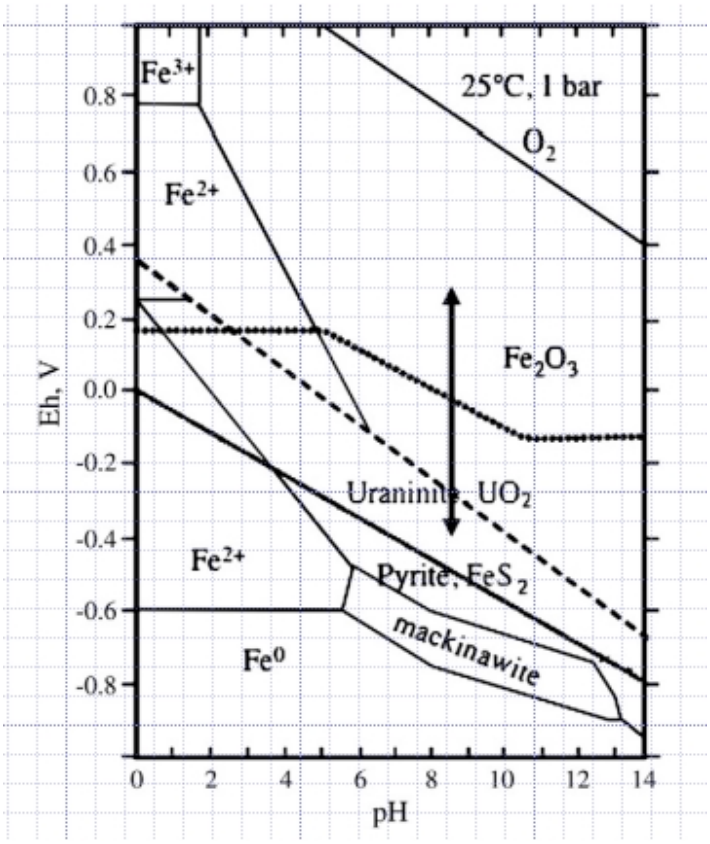


Figure B.12-1. Partial Eh-pH diagram Fe-U-S-O.

The bench scale system, which was operated successfully for 3.5 yrs is a 55 gal drum scale column as illustrated in Figure. B.12-2.

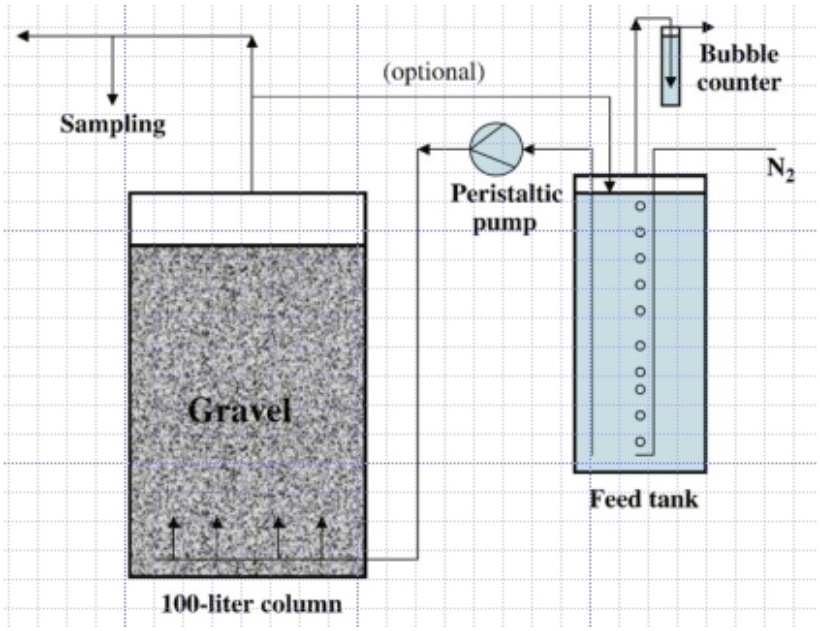


Figure B.12-2. Schematic of the column experiment.

B.12.3.2 Field Pilot Test

A confined field test was conducted adjacent to the tailings pond, which after bio-stimulation showed similarly encouraging results as in the laboratory. Taking Dänkritz 1 as an example we show that in situ bioremediation can be available option for long-term site remediation, if the process is designed based on sufficient laboratory and field data. The pilot test system is illustrated in Figure B.12-3.

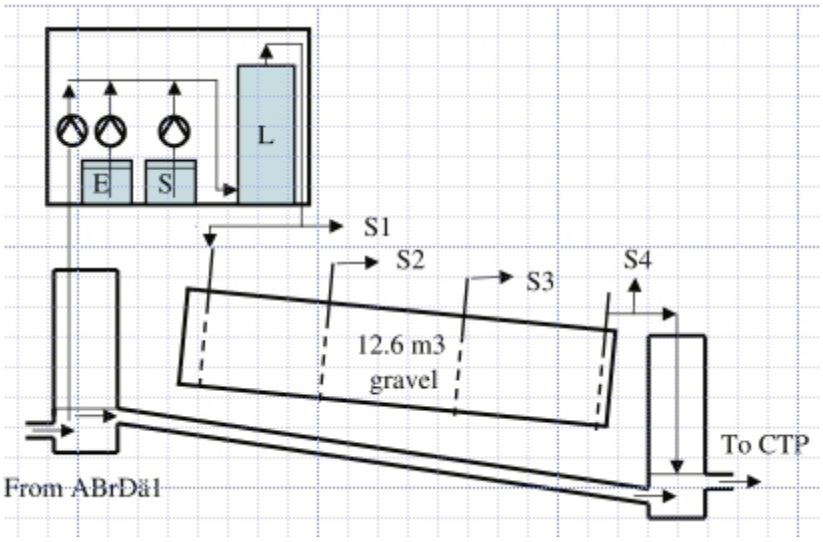


Figure B.12-3. Schematic representation of the confined field test. S1 to S4 = sample ports; E &S = reservoirs for ethanol, STMP, respectively, L = pre-reactor; CTP = Chemical treatment plant.

B.12.4 BCR Performance

The results of the confined field experiment (Figure B.12-4) can be interpreted by assuming that biotic reduction of uranium, iron, and sulfate is taking place and that uraninite and iron sulfide (mackinawite) precipitate on the gravel. These processes are relatively fast and effective. It is most likely a matter of optimizing the amount of amendment and potentially of additional amendments in individual chambers of the column to further lower the uranium concentration to a desirable level. Based on our results for uraninite stability, we conclude this discussion with a speculation about the effectiveness of in situ bioremediation and the stability of uraninite in the underground after termination of a respective process at Dänkritz 1.

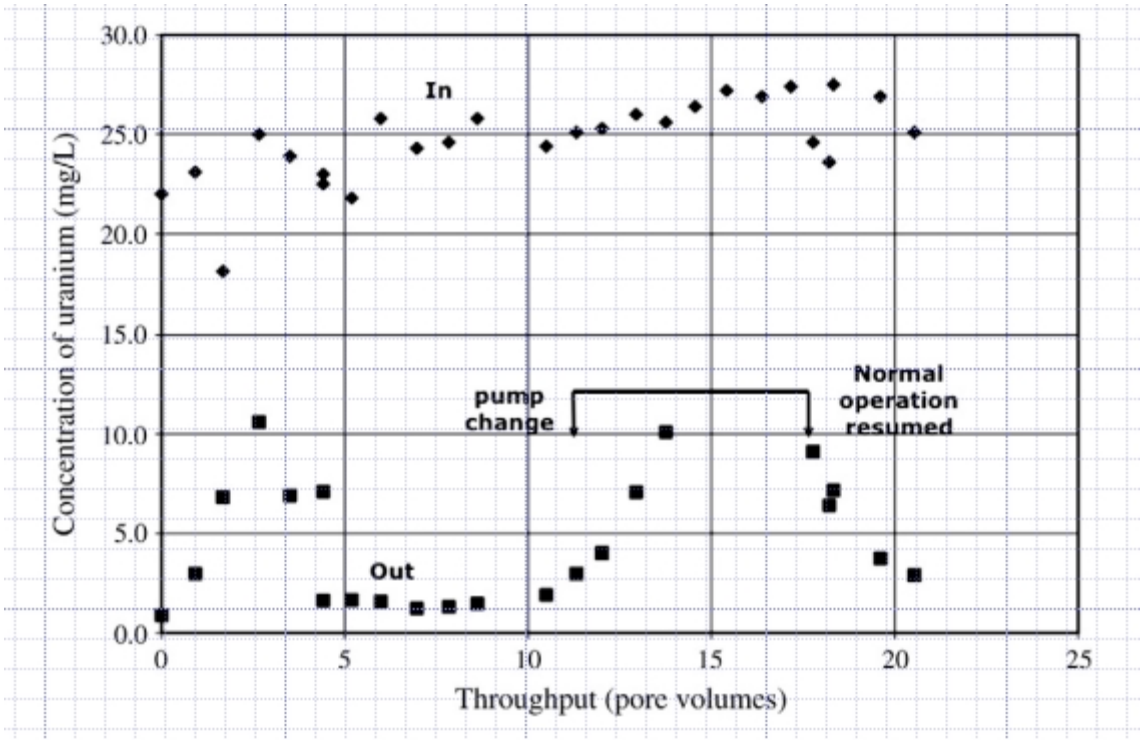


Figure B.12-4. Input and output concentration of uranium (pilot test)

B.12.5 Conclusions

Laboratory and field tests provided evidence that uranium in contaminated groundwater can be reduced, precipitated and accumulated as uraninite (UO_2) on soil surfaces by in situ bioremediation. Fe(III) in minerals such as goethite and hematite in the host rock is reduced biotically to Fe(II) and sulfate to sulfide. As a result, iron sulfide precipitates, e.g. as mackinawite (FeS 0.9). Iron sulfide acts as a redox buffer protecting uraninite from oxidation by in filtration of dissolved oxygen. The sulfide is less stable thermodynamically and is oxidized to sulfate. An atomic ratio S:U=18:1 was large enough to protect uraninite for 2.4 years in the presence of oxygen-saturated tap water. Only 1% of the deposited uraninite was released during this time and the uranium concentration was <35 µg/L. The results of this study support our hypothesis that uraninite can be stabilized by mackinawite and that an in situ bioremediation process can be developed based on this finding.

B.12.6 Regulatory and Stakeholder Challenges

This bioremediation process was strongly supported by the regulators and stakeholders because it was viewed as a natural biological system and, being on-site, did not produce secondary waste. These supporters felt this option was superior to pump and treat or to hydrological control of recharge water to the tailing piles. However, the system is a pilot scale test and is presently only

applicable to small seepages from the tailing piles. Therefore, bioremediation is only one of perhaps several remedial options.

B.12.7 Other Challenges and Lessons Learned

The pilot field test showed that changes in groundwater chemistry can negatively affect the bacteria's ability to directly reduce uranium^{+6}to uranium^{+4} and thus remediate the groundwater.

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B.13 Copper Basin Mine Case Study

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B.13.1 Name, Location, and Site Description

Copper Basin Mining Site is in the Lower Potato Creek Watershed near Ducktown, Polk County, in southeastern Tennessee near the North Carolina and Georgia borders. This site is approximately 50 square miles and lies adjacent to lands administered by the Cherokee National Forest (Figure 1-1). The Copper Basin, north of the Ocoee River, is drained by North Potato Creek and Davis Mill Creek, which empty into the Ocoee River.

The basin has been host to 150 years of copper mining, beneficiation and mineral processing, sulfuric acid, and other chemical production that have left a legacy of environmental degradation. Deep mining was the primary method, with a few surface-mining operations. In addition to mining and related operations, associated support infrastructure (railroads, equipment storage, and other waste processing facilities) can be found throughout the basin. Remnants of these mining practices remain scattered throughout portions of both North Potato Creek and Davis Mill Creek watersheds. The affected media are soil, sediment, surface stream water, surface pool water, and groundwater.

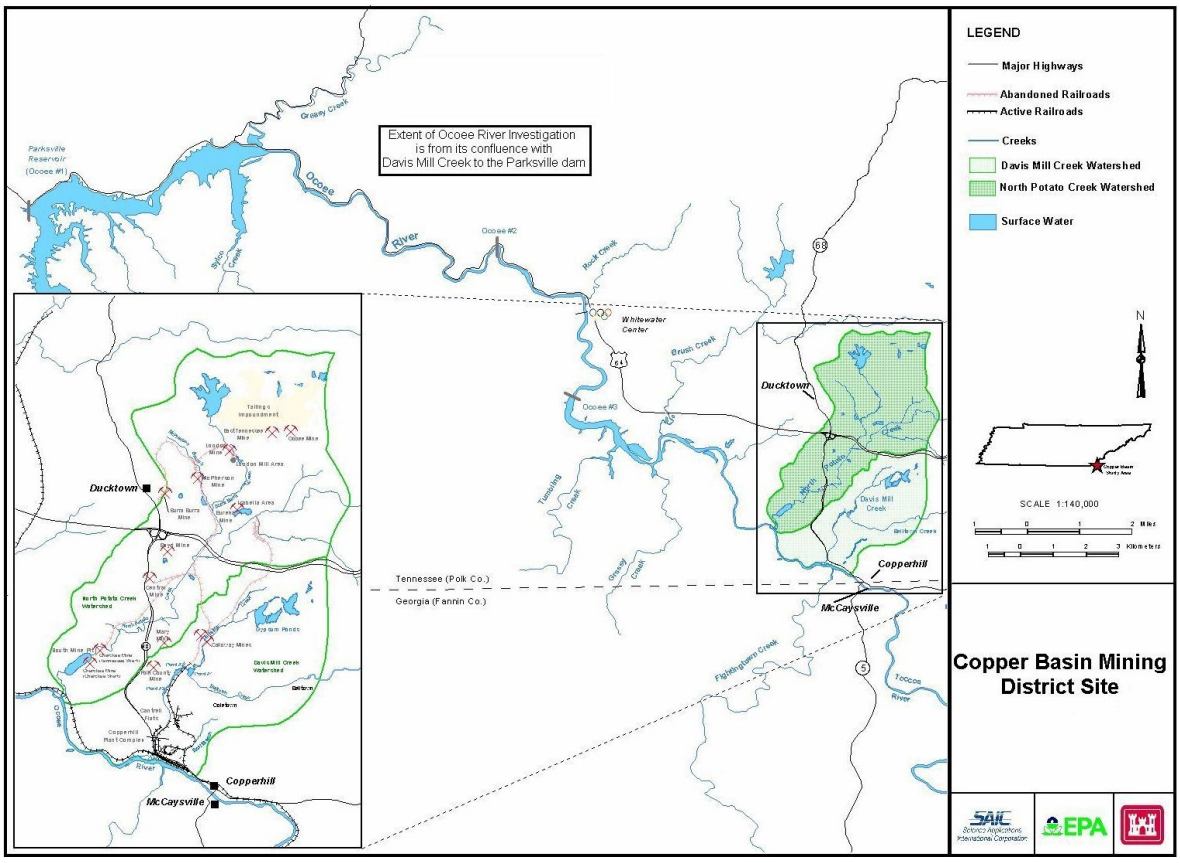


Figure B.13-1. Copper Basin Mining District.

Before remediation efforts, the site contained extensive acidic drainage with aluminum, copper, iron, manganese and zinc as the primary contaminants. Secondary contaminants include arsenic, cadmium, cobalt, copper, lead, mercury, selenium, and sulfate. Remedial actions at the Copper Basin Project by Glenn Springs Holdings, Inc. (GSH) are regulated by USEPA Region 4 Superfund and Tennessee Department of Environment and Conservation under its Superfund Voluntary Cleanup Oversight and Assistance Program. The site cleanup goal is mitigation of human health risk and ecological risk in the stream environment and is evaluated using load reductions in contaminants in selected streams, supplemented by performance goals of establishment of diverse benthic macroinvertebrates in North Potato Creek, establishment of vegetative cover on reclaimed waste areas, and establishment of fish and amphibians in wetlands in certain water bodies at the headwaters of North Potato Creek. Lower North Potato Creek watershed restoration has been addressed using the adaptive management process. Adaptive management involves well-reasoned iterative actions accompanied by comprehensive monitoring to assess progress toward quantitative objectives and facilitate subsequent remedial decisions. Specifically, GSH has sequentially 1) characterized drainage and influences, 2) diverted unaffected drainage, 3) captured and chemically treated affected drainage, 4) sequestered acid producing materials, and 5) mitigated remaining problems with passive systems (both wetlands and biochemical reactors).

In 1998, GSH removed obvious waste material proximate to a first order stream and diverted the base flow from the stream via a concrete dam to a two acre anaerobic wetland outside the flood plain. This was necessary since the vegetative cover in the watershed had not yet achieved stability and erosion and sediment load during storms was significant. The wetland has treated base flow (~300 gpm) of McPherson Branch since that time and has removed all acidity, provided alkalinity, and substantially reduced iron, copper, manganese, and zinc. The wetland was supplemented with three aerobic wetlands and a manganese reducing rock filter in 2003. A restored stream segment was added to the drainage before it returned to the receiving stream as part of the demonstration effort to learn more about the challenges facing re-establishment of diverse aquatic life in the dramatically impacted habitat.

Reaches of some receiving streams failed to meet macroinvertebrate performance goals after diversion of affected drainage and removal of evident waste materials. Some reaches recovered after these efforts, but others required more extensive removal of tailings and waste materials and reconstruction of nearly one mile (1600 meters) of second order stream. During impaired stream sediment and waste removal efforts, all acid forming materials were removed from the site, and "compost vaults" were installed in moist areas where either waste materials or their influence had the potential to contribute poor quality drainage. These small biochemical reactors differed from the McPherson Demonstration Wetland in that post reclamation water quality within the reclaimed area (now interstitial groundwater seepage to the restored stream) was not characterized before or after the installation of the many compost vaults placed proximate to the restored stream banks. The efficacy of the remedial efforts can be attributed to the combined efforts of the five pronged sequential approach described in the first paragraph, but the specific contributions toward improving water quality from the compost vaults cannot be isolated or quantified. At other locations, waste was removed and compost vaults installed that drained to the receiving stream without substantial in-stream removals or habitat enhancement.

B.13.2 MIW Chemistry

Drainage to McPherson Branch from undisturbed metal laden strata and mine waste placed along transportation corridors and remaining at roasting yards and processing areas caused the stream to contain metals and associated acidity that exceeded established water quality criteria. The pH was steady at ~4.2, with ~ 30 mg/L acidity, and the total iron, manganese, copper, zinc, and aluminum concentrations were 7.0, 1.2, 0.6, 1.7, and 4.2 milligrams per liter (mg/L), respectively. This drainage was diverted to the McPherson Demonstration Wetland in 1998, and with the addition of aerobic wetlands and rock filters in 2003 to supplement the original two acre aerobic wetland, effluent drainage has since been alkaline with pH > 6.5 and all metals (with the exception of zinc during very cold weather) less than water quality criteria. Drainage to the receiving stream bypasses several hundred linear feet of stream that has since been excavated and restored with non-acid producing materials and supplemented with cobbles and graded substrate. Moist areas that might contain residual waste materials (or their influence) after removal of obvious sources within the stream corridor were excavated and "compost vaults" were installed. There was no focused characterization of the seepage as the construction of the stream channels and associated excavation occurred in the flood plain, within work areas isolated from gravity drainage by progressive

diversions and pumps. Accordingly, it has been impossible to characterize post construction water quality from the compost vaults as they are integrated into the interstitial drainage to the stream and have no defined outlet. Temporal stream water quality improvements since construction and non-impairment of water flowing proximate to these areas of the restored stream are measures of efficacy. In Ellis Branch, slag and other mine waste drained directly to the stream. Clay barriers were installed on the banks of the stream and extensive waste removals were performed well below the elevation of the undisturbed (but impacted) stream bed. The waste material was replaced with "compost vaults" and drainage from these subsurface structures was directed to the receiving steam at a specific interval, and contributes an unmeasured, uncharacterized flow to the stream. No formal attempt to characterize the drainage after removal or reclamation was possible, as all drainage was subsurface after construction. Temporal stream water quality improvements since construction and non-impairment of water flowing proximate to these areas of the undisturbed stream are measures of efficacy.

B.13.3 System Design

The McPherson Demonstration Wetland was designed for flexible treatment of the base flow of McPherson Branch. Sediment laden storm water (generally with lower concentrations of metals) was allowed to bypass the wetland. This prevented inundating the wetland and associated desirable subsurface reactions with sediment. A pinch valve in a pipe through the dam regulates the water level behind the dam and into the wetland. A Geotextile Clay Liner was placed over waste material and associated native underlying alluvium and a two acre anaerobic wetland was constructed that provided for alternate flow paths through the system.

Flow from the stream enters a rock pool at the upstream end of the system, and flows through the limestone bed under the humic strata (square hay bales overlain with mushroom compost). The flow is collected at the downstream end of the system by a PE pipe header and directed to an Agridrain® level control structure. An Agridrain® structure regulates the subsurface flow through the crushed limestone aggregate layer and overlying hay and mushroom compost layers (humic strata). Alternately, the subsurface flow may be reduced by adding stop logs in the Agridrain® and surface flow over the humic strata and planted Typha (cattails) is regulated by an overflow structure. Concrete traffic barriers atop the limestone layer with notches force a serpentine surface flow through the wetland, increasing residence time. Since installation, the flow has typically been subsurface, but the overflow serves to provide a safety measure during unexpected precipitation events. Regulating the water level in the system is accomplished by daily monitoring and occasional adjustment of the pinch valve and the stop logs in the Agridrain®. Figures B.13-2 and B.13-3 illustrate the layout and cross sections of the anaerobic wetland follow.

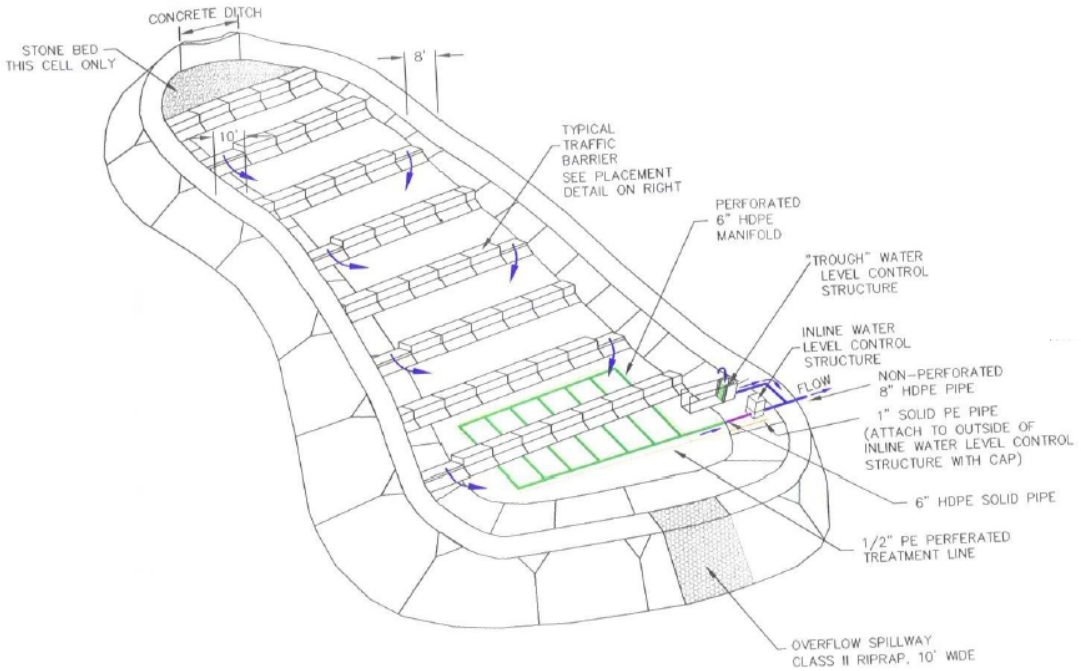


Figure B.13-2. Anaerobic wetland layout.

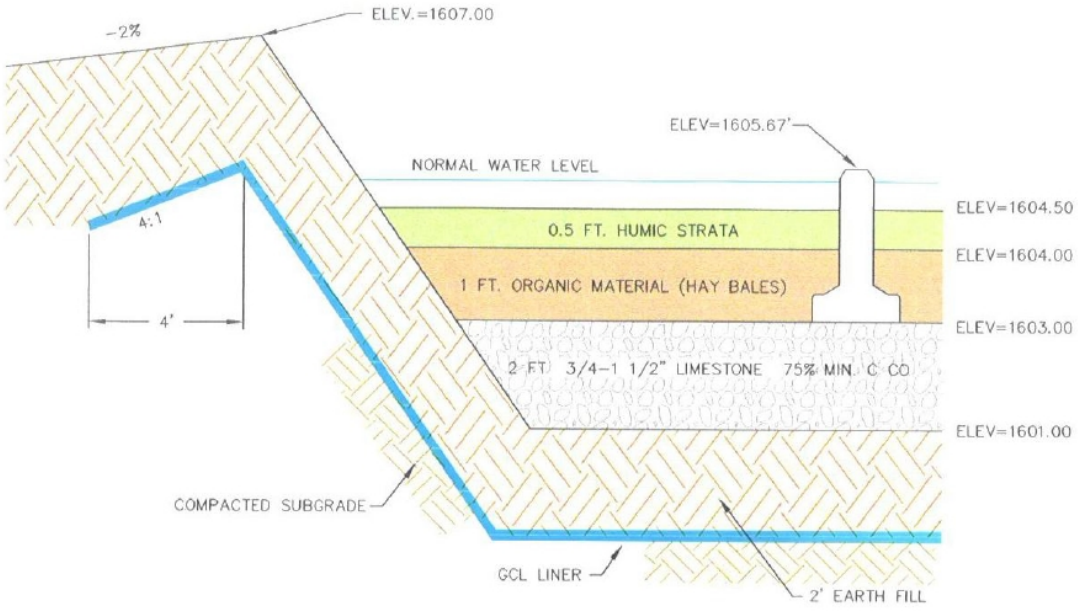


Figure B.13-3. Anaerobic wetland cross-section.

Since chemical reactions augmented by bacterial populations within the limestone bed and humic strata are the mechanisms of contaminant removal, this system is, by definition, a biochemical reactor. The "polishing" wetlands that follow the system are aerobic, as the oxygen deficient effluent from the anaerobic wetland flows over and through the cattail dominated pools. All or a portion of the flow is then introduced subsurface to an un-vegetated bed of crushed limestone and collected in a pipe header where it joins any untreated effluent from the aerobic wetland and discharges to the receiving stream. This "rock filter" removes manganese and other metals from the drainage.

The "compost vaults" are more subtle passive treatment systems and are placed along stream corridors where mine waste removal has been completed. Compost vaults are irregular pits dug with an excavator in selected areas near stream beds that may be impacted by remaining waste materials. In areas where the excavation remains moist (potential groundwater sources), the area is over-excavated and a layer of crushed limestone aggregate is overlain with a layer of spent mushroom compost. This combination of materials was determined to be effective in ameliorating site specific contaminants in extensive static and dynamic bench scale testing. The vault collects drainage, removes and sequesters metals, and conveys it downgrade to the hyporheic zone and ultimately the stream channel. Typical x-sections follow.

B.13.4 BCR Performance

McPherson Wetland performance has been tracked since August 2003. The performance indicators are key metals as well as acidity, sulfate and pH. Table B.13-1 provides a summary of average influent and effluent (from each component) over the period between August 2003 and February 2012.

Table B.13-1. Summary of average influent and effluent concentrations.

	Anaerobic Influent	**Anaerobic Effluent**	**Aerobic Effluent**	**Rock Filter Effluent**	**System Outlet**
Parameter	402	403W	406	420	440/440A
Aluminum (dissolved)	764	29.3	19.9	21.3	16.5
Copper (dissolved)	136	5.6	3.2	2.2	3.3
Iron	203	992	567	78.8	153
Iron (dissolved)	136	746	86.6	24.9	20.5
Manganese (dissolved)	1231	1101	867	54.5	188
Zinc (dissolved)	591	242	327	63.3	130
Acidity	7.9	3.9	NM	NM	0.7
Sulfate	124	109	98	108	116
pH	4.9	7.0	7.0	7.2	7.2
Notes: Metals units = µg/L; Acidity and sulfate = mg/L					

The following provides detailed discussions of the treatment performance over time for each performance indicator.

B.13.4.1 Dissolved Aluminum

- 402 to 403w

  The anaerobic unit (402 to 403w) removed dissolved aluminum on average over the monitoring period (mean 47%/median 97%) with none of 58 403w results exceeding 100 µg/L. No seasonal patterns were evident in the removal efficiency evaluation.

  Prior to late 2006, removal efficiencies were consistently above 90%. Influent aluminum concentrations dropped from over 1 mg/L to less than 0.3 mg/L after completion of stream reconstruction downstream of the former McPherson Branch railroad wye in late 2006. Correspondingly, removal efficiency also declined and became more variable with occasional minor additions observed. The pattern was attributable in part to reduced influent concentrations and effluent results often bounded by the detection limit. Average influent dissolved aluminum concentrations trended up after late 2009 (possibly associated with decreased in-stream treatment above 402), and removal efficiency also rebounded.

- Overall

  The overall acid mine drainage AMD demonstration wetland system consistently removed dissolved aluminum over the monitoring period (mean 75%/median 98%) with none of 63 effluent results exceeding 100 µg/L. No seasonal patterns were evident in the removal efficiency evaluation.

  Until late 2006, removal efficiencies were consistently above 95%. After that time, removal was more variable although it remained above 75% on average since December 2008 with one exception. The pattern was attributable in part to reduced influent concentrations and effluent results often bounded by the detection limit.

B.13.4.2 Dissolved Copper

- 402 to 403w
- The anaerobic unit (402 to 403w) removed dissolved copper on average over the monitoring period (mean 96%/median 98%), and in only three instances was the efficiency less than 90% (minimum 62%). Seasonal patterns were evident prior to December 2006 with lower removal efficiency during winter months. During that period, 403w concentrations ranged up to 64 µg/L.

In late 2006, influent dissolved copper concentrations dropped from an average of around 200 µg/L to below 100 µg/L after completion of stream reconstruction downstream of the former McPherson Branch railroad wye. From that time forward, removal efficiency was consistent (above 90%) with no seasonal changes.

Overall

The overall AMD demonstration wetland system consistently removed dissolved copper over the monitoring period (mean 97%/median 98%). Seasonal and temporal patterns for the overall system were comparable to those observed for the anaerobic unit although some limited additional removal was achieved in downstream units. The effluent dissolved copper concentration did not exceed 10 µg/L after December 2006 and removal efficiency exceeded 90% except when influent concentrations were notably low.

B.13.4.3 Total Iron

- 402 to 403w

  The anaerobic unit (402 to 403w) consistently added total iron over the monitoring period. The amount of addition dropped during the winter months and the discharge (403w) concentration was often above 1000 µg/L.

  The seasonal pattern is thought to be associated with enhanced reducing conditions (iron solubilized) when influent and ambient temperatures were higher during the summer months.

- Overall

  The overall AMD demonstration wetland system generally provided some net removal of total iron (mean 35%/median 37%) although the overall system occasionally had some net addition. Overall total iron removal also exhibited a seasonal pattern although less prominent than observed in the anaerobic unit alone. Effluent concentrations did not exceed 1000 µg/L, and were typically less than 300 µg/L.

  Positive average removal rates and effluent concentrations indicate that the aerobic and/or rock filter units provided some total iron removal benefit.

B.13.4.4 Dissolved Iron

402 to 403w

The anaerobic unit (402 to 403w) added dissolved iron with few exceptions over the monitoring period. Similar to total iron, the amount of addition dropped during the winter months and the discharge (403w) concentration was often above 1000 µg/L.

The seasonal pattern is thought to be associated with enhanced reducing conditions (iron solubilized) when influent and ambient temperatures were higher during the summer months. Influent acidity and iron dropped in late 2006 after completion of stream reconstruction downstream of the former McPherson Branch railroad wye. After that time, the removal efficiency also declined although this was attributable in part to reduced influent concentrations and effluent results were often bounded by the detection limit.

Overall

The overall AMD demonstration wetland system generally provided some net removal of dissolved iron (mean 64%/median 83%). No obvious seasonality was observed. Effluent concentrations were consistently at or below 100 µg/L over the monitoring period.

Prior to late 2006, the overall system consistently reduced dissolved iron. After December 2006, however, the removal efficiency decreased although this was attributable in part to reduced influent concentrations and effluent results often bounded by the detection limit. When influent concentrations rose in early 2011, overall dissolved iron removal also rebounded. Positive average removal rates and effluent concentrations indicate that the aerobic and/or rock filter units provided some dissolved iron removal benefit.

B.13.4.5 Dissolved Manganese

402 to 403w

Dissolved manganese removal in the anaerobic unit (402 to 403w) was with few exceptions below 50% and frequently less than 25% (average 3%/median 9%). Prior to June 2007, dissolved manganese additions were fairly common in the unit. After that time, however, some removal did occur; generally in excess of 40%.

Seasonal trends were apparent prior to May 2007 with higher removal rates during warmer periods. Influent dissolved manganese concentrations did not change appreciably (on average) after completion of the stream reconstruction downstream of the former McPherson Branch railroad wye but did show greater variability.

Overall

The overall AMD demonstration wetland system consistently removed dissolved manganese over the monitoring period (mean 83%/median 94%). Seasonal trends were apparent prior to 2007 with lower removal rates (less than 30%) during winter months bottoming out in March 2005 when a slight addition was indicated. From 2007 to early 2012, however, little seasonal variability was observed and overall removal rates were sustained at or above 80% throughout the year. It should be noted that only a fraction of the total flow was routed to the rock filter under most circumstances; controlled by the unit's hydraulic capacity. Overall manganese removal efficiency was heavily dependent upon flow distribution with higher rates evident during periods of relatively low influent flow.

B.13.4.6 Dissolved Zinc

402 to 403w

The anaerobic unit (402 to 403w) removed dissolved zinc on average over the monitoring period (mean 59%/median 86%). Removal rates were typically greater than 90% although exceptions were noted during winter months. Minimum removal rates were usually observed during January or February; typically less than 25% with negative values (additions) common. During late spring

through early fall, effluent (403w) concentrations were consistently below 100 µg/L but ranged as high as 1000 µg/L during the colder months.

From June 2007 to September 2009, the cold weather pattern was not observed and removal rates were consistently above 85%. That period was characterized by lower average influent flow and presumably higher anaerobic unit retention times, and immediately followed completion of stream reconstruction downstream of the former McPherson Branch railroad wye. Influent zinc concentrations have been lower on average since stream reconstruction albeit variable between sampling events.

Overall

Dissolved zinc removal patterns for the overall AMD demonstration wetland system were similar to those observed for the anaerobic unit. The removal rate was higher (average 77%/median 90%) and seasonal fluctuations were less extreme. Net addition for the overall system was observed during winter 2006, 2007 and 2010 only. From June 2007 to September 2009, the cold weather pattern was not observed and removal rates were consistently above 90%. The sustained high over winter removal rates corresponded with the period of lower average influent flow discussed above.

The patterns indicated additional removal after the anaerobic unit attributable to the aerobic and rock filter units.

The seasonal metals/acidity/sulfate/dissolved oxygen (DO) removal, ORP and alkalinity addition patterns could be attributable to colder temperatures (ambient and influent) which would affect bacterial activity and/or flow as it peaked during the winter months. For the latter, there does appear to be some correlation between peak flow and the magnitude of performance drops.

B.13.4.7 Acidity

The evaluation of acidity removal in the AMD Demo wetlands was based upon calculated values. Laboratory acidity results were found to be inconsistent and data quality checks identified potential discrepancies in many instances. Calculated acidity data are presented in Table 7.1.

402 to 403w

The anaerobic unit (402 to 403w) removed acidity on average (average 46%/median 56%) over the monitoring period. Prior to September 2006, the acidity removal rate consistently exceeded 50%. Upon completion of the stream reconstruction downstream of the former McPherson Branch railroad wye in late 2006, influent acidity concentrations abruptly dropped by 50% restabilizing in the 3 to 5 mg/L range. Over the next six months, small acidity additions were indicated. From September 2007 to early 2012, acidity removal was highly variable (but always positive) with maximum rates during winter months.

Overall

The overall AMD demonstration wetland system removed over 80% of influent acidity with few exceptions (average 91%/median 93%). The lowest removal rates (as low as 68%) were observed during the winters of 2004 and 2005 corresponding with maximum flow rates through the system. After September 2006, effluent acidity concentrations were consistently below 1.2 mg/L. The patterns indicated that the aerobic and rock filter units were vital for effective acidity removal.

B.13.4.8 Sulfate

- 402 to 403w

  Sulfate removal in the anaerobic unit (402 to 403w) was less than 20% (average 6%/median 4%) with few exceptions. Seasonal trends were observed with maximum removal during summer months. Sulfate additions were commonly observed during winter months. The maximum removal occurred during the summers of 2007 and 2008 (up to 45%) and corresponded with the lowest influent flows over the monitoring period.

- Overall

  The overall AMD demonstration wetland system sulfate removal rates and patterns were nearly identical to those observed for the anaerobic unit. There was little indication that the aerobic or rock filter units provided any appreciable sulfate removal.

B.13.4.9 pH

- 402 to 403w

  The anaerobic unit (402 to 403w) consistently raised pH from an average of 4.9 s.u. (median 4.5 s.u.) to 7.0 s.u. (median 7.0 s.u.). No seasonal trends were observed.

  In late 2006, influent pH rose from the 3.5 to 4.5 s.u. range to above 5 s.u. (with few exceptions) after completion of stream reconstruction downstream of the former McPherson Branch railroad wye in late 2006. Prior to September 2006, the pH change through the unit averaged around 2 s.u., but was closer to 1 s.u. afterward.

- Overall

  The overall AMD demonstration wetland system exhibited patterns comparable to those observed for the anaerobic unit. During most events, overall system effluent pH was slightly higher (average 7.2 s.u./median 7.4 s.u.) than that observed at 403w indicating additional benefits were provided by the aerobic and rock filter units.

B.13.5 BCR Monitoring

The McPherson Demonstration Wetland monitoring program was initiated after approximately 3 weeks of stabilization and implementation continues with approved modifications Key monitoring

locations are:

- 402 - Influent to the anaerobic wetland
- 403w - Effluent from the anaerobic wetland and influent to the aerobic wetland
- 406 - Effluent from the third aerobic detention cell and influent to the rock filter
- 420 - Effluent from the rock filter and influent to RSS
- 440/440A - System outlet

Between August 2003 and February 2012, monitoring efforts included:

- Daily monitoring of water levels in the anaerobic wetland and diversion dam
- Weekly field monitoring of flow, temperature, pH, specific conductance, DO concentration and % saturation, and oxidation/reduction potential (ORP) for the first year. Changed to monthly in August 2004 with the exception of water levels (flow) and pH which are still measured weekly. Further reduction in sampling frequency was approved in 2007, 2008 and 2009.
- Biweekly field analysis of manganese and sulfide for the first year.
- Monthly laboratory analysis of the following parameters the first year: pH, alkalinity, acidity, sulfate, hardness, biochemical oxygen demand (BOD_5), total organic carbon (TOC), total suspended solids(TSS), total dissolved solids (TDS), aluminum, calcium, cadmium, cobalt, copper, iron, lead, manganese, selenium, sodium, and zinc. Metals are analyzed for both total and dissolved concentrations. Request approved in August 2004 to eliminate BOD_5, TOC, TDS, and selenium from the monthly analysis for all locations and eliminate other parameters for certain locations. This was eliminated by subsequent monitoring revisions. These field parameters were included in the quarterly and semi-annual monitoring efforts below.
- Quarterly laboratory analysis of wetland inlet (402) and system outlet (440/440A) initiated in March 2009
- Semi-annual laboratory analysis of anaerobic outlet (403w) initiated January 2010 per TDEC approval.
- 402 and outlet as part of BBC WS study since June 2011

Currently, monitoring at the McPherson Demonstration wetland includes:

- Daily monitoring of water levels in the anaerobic wetland and diversion dam
- Weekly field monitoring of flow and pH at all points
- Quarterly (March, June, September, December) laboratory analysis of wetland inlet (402) and system outlet
- Field monitoring and laboratory analysis of 402, 403W and outlet (440/440A) during January and July.

B.13.6 Regulatory Challenges

Due to the unique regulatory framework guiding the remedial programs in the Lower North Potato Creek watershed, there have been relatively few challenges. Since initial agreements were reached in January 2001, GSHI, USEPA, TDEC and project contractors have held bimonthly technical status meetings. This forum has been used to present technical data, discuss strategies for remedial actions and resolve issues associated with potential conflicts between adaptive management –based decisions and regulatory requirements. As a result, all parties have had the opportunity to voice opinions and provide technical input while maintaining perspective on the merits of individual actions (such as passive treatment) in light of overall projective objectives.

B.13.7 Stakeholder Challenges

As mentioned above, regulatory stakeholders have had a voice throughout the process. The local community has been kept informed of activities through a series of public meetings. GSHI is well-regarded in the community.

B.13.8 References

Faulkner, Ben B., Miller, Franklin. 2010. Land and Water Reclamation of the Copper Basin of TN in 2010 Symposium Proceedings American Society for Mining and Reclamation, Pittsburgh,PA http://www.asmr.us/Publications/Conference%20Proceedings/Conference%20Proceedings.htm

Miller, Franklin, Faulkner, Ben. 2008. Land and Water Reclamation of the Copper Basin of TN in Proceedings of USEPA/National Ground Water Association Workshop, Denver, CO. http://ngwa.confex.com/ngwa/mine08/webprogram/start.html

Faulkner, Ben B. et al. 2007. Bench-Scale Testing of a Sulfate-Reducing Bacteria Treatment of Copper Basin, TN Mine Drainage in WVDM Task Force Symposium Proceedings. http://www.wvmdtaskforce.com/proceedings/07/Faulkner-2007-TF.pdf

B.14 Lady Leith Mine Case Study

Acknowledgments

The mining team would like to acknowledge James Gusek of Sovereign Consulting Inc., Lakewood, CO, who submitted the Lady Leith Mine Biochemical Reactors Case Study.

B.14.1 Site Information

Contacts

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Name, Location, and Site Description

The Lady Leith Mine is located in Jefferson County, Montana within Deerlodge National Forest. The site is administered by the US Department of Agriculture, Forest Service (US Forest Service). The BCR system was constructed by Sonle Technical Services under contract to Envirocon, Inc. Golder designed the system based on data provided by Sonle.

The treatment system is designed to remove iron and zinc from mine influenced water (MIW) collected from the Lady Leith Mine; it consists of a single sulfate reducing bioreactor (SRBR) or BCR fed from the bottom as shown in the Drawings. The BCR was installed/retrofitted into a preliminary excavation developed by Envirocon in 2006 in the vicinity of the collapsed Lady Leith portal; this site is adjacent to a perennial un-named stream and the site access road (see Figure B.14-1).

From available information at the time, it appeared that a soil berm had been installed to temporarily protect the excavation/work area from stream flooding. As of June 21, 2007, the excavation appeared to be flooded as a result of local groundwater infiltration; the Lady Leith MIW was temporarily diverted to the stream.

B.14.2 MIW Chemistry

The Lady Leith design MIW (data from Sonle) was net neutral (pH 8.0 to 8.3) with minor amounts of iron (0.1 mg/L) and zinc (0.3 mg/L). Sulfate of 42 mg/L was considered borderline low for a sulfate reducing BCR. Actual values of iron and zinc were slightly above these concentrations (see Table 2), but not detrimentally so. Actual influent pH was about 6.6 s.u (see Table 1).

Table B.14-1. Field parameter measurements, Lady Leith Mine passive treatment system

		Samples Collected 9/14/07			Samples collected 10/1/2007			Samples Collected on 10-12-07		
Analyte	Units	LLM INF	LLM PW	LLM APZ	LLM INF	LLM PW	LLM APZ	LLM INF	LLM PW	LLM APZ
pH	s.u.	6.6	6.94	7.06	6.6	6.94	7.06	6.41	6.92	7.13
Temperature	°C	5.6	4.4	5.8	5.6	4.4	5.8	4.3	3.6	4.0
Conductivity	ms/cm	318	384	376	318	318	376	327	310	311
ORP	mV	233	225	245	233	233	245	266	257	272

Notes
Field parameters were measured by Sonle during sample collection activities
s.u. = standard units
ms/cm = milliseimens per centimeter
mV = millivolts
LLM INF = Influent Sampling Location
LLM PW = Ponded Water Sampling Location
LLM APZ = Aerobic Polishing Zone Effluent Sampling Location

Table B.14-2. Summary of analytical laboratory results, Lady Leith Mine passive treatment system.

		DL	Samples Collected 9/14/07			Samples collected 10/1/2007			Samples Collected on 10-12-07		
Analyte	Units		LLM INF	LLM PW	LLM APZ	LLM INF	LLM PW	LLM APZ	LLM INF	LLM PW	LLM APZ
Silver	mg/L	0.0024	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Aluminum	mg/L	0.018	0.032	BDL	0.099	BDL	BDL	BDL	BDL	0.049	0.051
Arsenic	mg/L	0.032	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Calcium	mg/L	0.01	28	34	33	31	40	41	29	36	36
Cadmium	mg/L	0.0009	0.0019	0.0031	0.0027	0.0044	0.0030	0.0038	0.0053	0.0034	0.0026
Cobalt	mg/L	0.005	0.0094	0.0054	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Chromium	mg/L	0.003	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Copper	mg/L	0.0009	BDL	0.0015	BDL	0.13	BDL	0.039	0.0050	0.0015	0.0015
Iron	mg/L	0.0021	0.81	0.39	0.30	1.6	0.69	0.71	0.91	0.46	0.51
Magnesium	mg/L	0.0003	6.7	9.9	9.8	7.6	11	11	7.0	9.2	9.5
Manganese	mg/L	0.0003	1.7	1.5	1.4	2.1	2.4	2.1	2.0	1.2	1.3
Lead	mg/L	0.02	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL	BDL
Sulfur	mg/L	0.11	24	17	15	17	15	15	16	15	15
Sulfate (Calc'd)	mg/L	0.33	73	50	46	52	44	45	48	46	45
Zinc	mg/L	0.0006	0.75	0.23	0.21	0.93	0.23	0.20	0.78	0.20	0.17

Notes
BDL = Target analyte was not detected at concentrations above the laboratory detection limit
mg/L = milligrams per liter
LLM INF = Influent Sampling Location
LLM PW = Ponded Water Sampling Location
LLM APZ = Aerobic Polishing Zone Effluent Sampling Location

B.14.3 BCR Design

The BCR was designed to treat approximately six (6) gallons per minute (gpm) of Lady Leith MIW; flows exceeding this amount were designed to be by-passed and mixed with treated water on the surface of the BCR. The BCR was fully lined with geomembrane and constructed to appear like a natural fen, or bottom-fed wetland. This approach was required to minimize damage from natural and human vandalism and assists in preserving the MIW temperature to improve bacterial activity in the system during the winter.

Two passive technologies were used to treat the Lady Leith MIW:

- The BCR (SRBR) with a total bottom area of about 210 square feet was built in the area just down gradient from the Lady Leith Mine portal. The SRBR cell was filled to a depth of

about 5.5 feet with a mixture of chipped wood, limestone sand, bacterial inoculum, and hay.

The treated MIW that ponded on the SRBR surface, containing biochemical oxygen demand (BOD) in addition to depressed dissolved oxygen levels, would flow by gravity across the SRBR surface to a spillway and thence into the receiving stream. The depth to the base of excavation may be as much as 7.5 feet; re-contouring may be required to achieve the recommended cell dimensions.

- An aeration polishing zone (APZ) on the surface of the SRBR functions as a free water surface (FWS) wetland for the aeration of SRBR effluent and the removal of excess BOD.

  MIW was to be collected in a bed of non-biodegradable Styrofoam shipping peanuts or other light-weight and inert material mixed with wood chips installed in the floor of the Lady Leith Mine collapsed portal; field fitting of this design feature was anticipated. The MIW was to be conveyed via a buried six-inch diameter high density polyethylene (HDPE) pipes to a buried flow control/diversion vault equipped with two four-inch diameter HDPE pipes; one pipe will direct feed water to the bottom of the SRBR, the other will direct overflow to the surface of the SRBR. Flow control valves and clean-outs included in the design would facilitate periodic major maintenance (estimated to be on the order of several decades) to be performed. All flows were to be by gravity.

See Section B.14.10, Design Drawings for design details. Details of the substrate mixture follow.

Table B.14-3. Approximate substrate amounts

Component	Weight Recipe	Approximate Unit Weights		Order Vol. Units[1]	Volume Units	Order Weight Units[1]	Weight Units
Inoculum (mostly manure)	10.0%	37	PCF	5.3	CY	2.6	tons
Chipped Wood A	10.0%	14.2	PCF	13.6	CY	2.6	tons
Chipped Wood B	40.0%	10.6	PCF	73	CY	10.5	tons
Limestone Sand	30.0%	95.6	PCF	6.1	CY	7.8	tons
Hay	10.0%	80	Lbs/bale	62	Bales	2.6	tons
Total	100.0%						tons

[1] Order quantities include a 5% safety factor
CY = Cubic Yards

Chipped Wood A and B is comprised of hardwood and softwood particles generated from trees or other acceptable sources. Wood Chip A may be comprised of chips that are typically angular in shape to stringy, linear shapes generated by a grinding action in the chipping machinery. Wood Chip B is a fine-grained type of chipped wood.

Pre-construction photos follow.





Source MIW



2006 Excavation by Envirocon



2006 Excavation by Envirocon



2006 Excavation by Envirocon

Construction photos follow.



B.14.3.1 Pre- and Posttreatment Requirements

There were no pretreatment steps. However, keeping the MIW anoxic in the capture zone was a design goal (to minimize iron oxy-hydroxide formation and plugging.

The APZ, on the top of the BCR, was to serve as a polishing step for BOD and excess iron.

B.14.4 BCR Performance

See Tables B.14-1 and B.14-2, above. There were three post-construction sampling events in 2007 to evaluate startup. Unfortunately, monitoring was suspended following these events and performance cannot be determined.

B.14.5 BCR Monitoring

Other than three startup monitoring events in 2007 referenced above, it is unknown if the Lady Leith BCR has been monitored or even visited since then. The team visited the site in July of 2012 and found that the influent water transfer pipe had plugged (potentially from scaling). The influent was flowing on the exterior of the pipe (a) and over the flow control/diversion vault (b). The team was unsuccessful in repairing/unplugging the influent flow pipe so adjustments were made to divert the surface flow onto the geomembrane, thus entering the BCR from the top (c). Immediately following diversion of the influent into the surface of the BCR iron precipitation began as can be seen in photo (c).



B.14.6 Regulatory Challenges

Unknown.

B.14.7 Stakeholder Challenges

Unknown.

B.14.8 Other Challenges and Lessons Learned

If indeed, the system has not been visited or maintained since construction, the credo that passive treatment is low-maintenance, rather than no-maintenance, has apparently been ignored.

The upflow configuration was initially selected due to the cold climate at the site. Based on current experience, it would be better to bury the BCR with a top covering of plastic infiltration chambers typically used in septic systems.

B.14.9 References

Golder Associates, Inc., 2007. "Statement of Work and Construction Guidelines Lady Leith Mine Site Passive Treatment System"; prepared for: Sonle Technical Services P.O. Box 31; Ellis-ton, MT 59728

Metesh, John et al., 1994. "Montana Bureau of Mines and Geology Open-File Report No. 321, Abandoned-Inactive Mines Program, Deerlodge National Forest, Vol. 1, Basin Creek Drainage. Prepared for the U.S. Dept. of Agriculture; Forest Service-Region 1. April 1994.

B.15 Luttrell Site Case Study

B.15.1 Contacts

The remedial project manager for the Upper Tenmile Creek Superfund Site is Tillman McAdams, and for the Basin side of the site, Kristine Edwards. The site contractor is CDM Smith, and the project manager is David Shanight.

B.15.2 Name, Location, and Site Description

The Luttrell site receives waste rock from the Upper Tenmile Creek Superfund Site and the BCR treats the effluent from that waste rock repository. The Upper Tenmile Creek Mining Area site is located in the Rimini Mining District, southwest of Helena, Montana. This site consists of numerous abandoned and inactive hard-rock mine sites that produced gold, lead, zinc, and copper. Mining began in the district before 1870 and continued through the 1920s. Little mining has been performed there since the early 1930s. USEPA added the Upper Tenmile Creek Mining Area to the Superfund National Priorities List on October 22, 1999, due to mining waste problems in the 53 square-mile watershed.

The small historic mining community of Rimini is located within the Superfund site boundaries. The site includes the drainage basin of Tenmile Creek upstream of the Helena water treatment plant and includes tributaries that supply water to the plant's five intake pipelines. USEPA identified 150 individual mine sites within the watershed boundary, of which 70 have been prioritized for cleanup. Many of these mines are above the five drinking water intakes for the City of Helena, which supply about 50% of the city's water.

B.15.3 BCR Design

A schematic of the BCR is presented in Figure B.15-1. The cell has a bottom surface area of approximately 3,600 square feet. The substrate used in the BCR includes a mixture of manure, wood chips, and straw, along with coarse limestone gravel for support. The BCR was filled, and the bacteria were given a long adaptation period from October 2002 to June 2003. During this period, the initial filling was supplemented only by rain and snowmelt, and the BCR lost some water by evaporation.

Some of the repository leachate has been diverted from the effluent pipe of the repository and has created input flows from 1 to 2.5 gpm to the BCR over the life of the reactor. The BCR influent pipe enters at the top of the BCR and the water flows vertically downwards through the cell. Water at the bottom of the cell is collected in a series of perforated collector pipes that lead to the main drain pipe that discharges on the east side of the cell (into the storm water cell). During the summer treatment season, a line is also connected to the BCR effluent pipe, which allows water transfer to a 15,000 gallon underground storage tank. This line also allows water to be pumped into a truck or into the leachate pond, depending on varying volumes of snowmelt, stormwater, and leachate to

manage each season. The BCR effluent is eventually routed with other collected waters to be treated in the active water treatment plant and is not directly discharged.

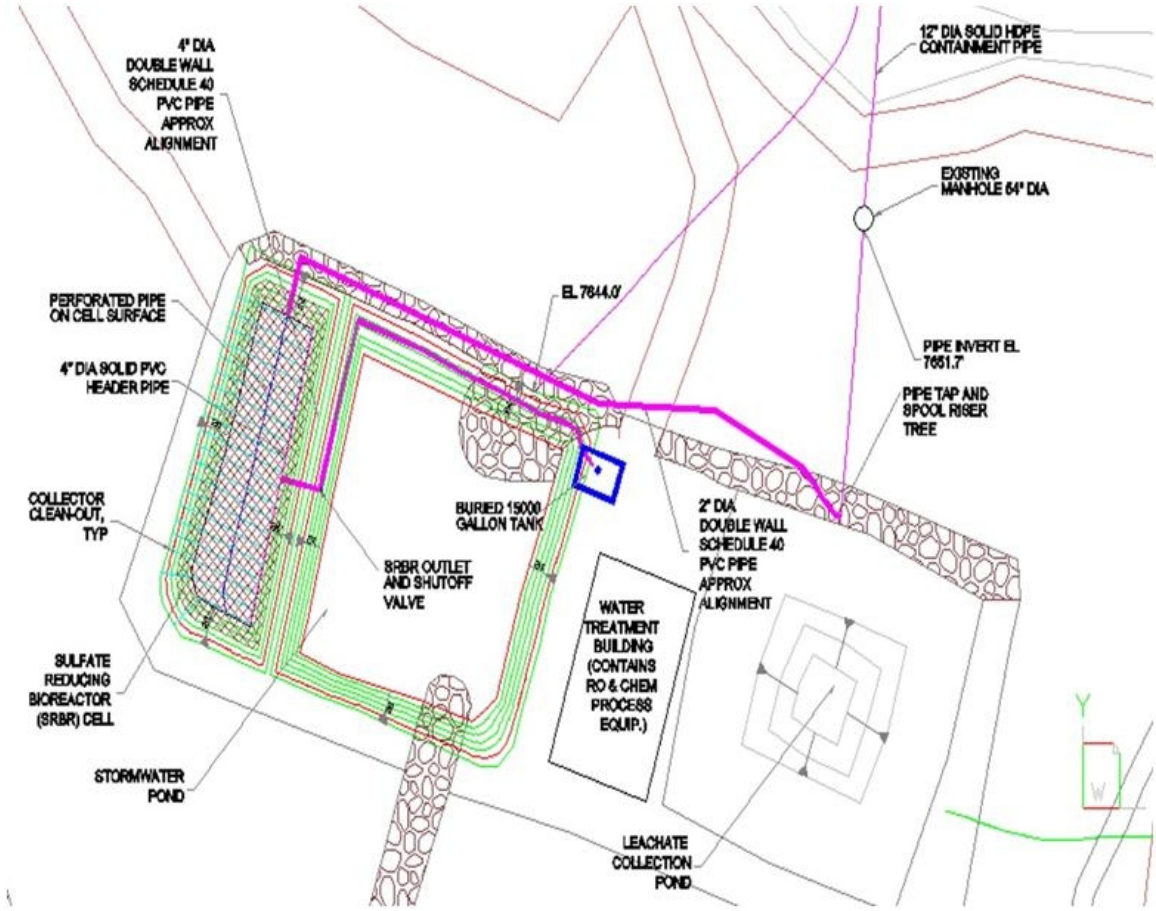


Figure B.15-1. Schematic of the Luttrell BCR.

The BCR has no flow in or out during the winter months (generally October through May) because snowfall and the lack of permanent buried plumbing restrict site access. In the winter, repository leachate drains to a leachate collection pond, where it is held for treatment the next season. Clean snowmelt that accumulates on top of the BCR is pumped into the adjacent stormwater pond.

B.15.4 BCR Operation and Data Collection

The BCR was filled and the bacteria were given a long adaptation period from October 2002 to June 2003. During this period, the initial filling was supplemented only by rain and snowmelt, and the BCR lost some water by evaporation. In 2005 and 2006, a more thorough study of the BCR effluent was undertaken to assist Region 8 in understanding how to use the BCR effluent for different mixing, while demonstrating the longevity of passive treatment for the repository leachate. This testing included various options of mixing either (1) with the treatment plant discharge and sending to the land application devices for discharge, or (2) mixing with the leachate to raise the

pH of the leachate and reduce the chemical costs of treating the leachate in the chemical precipitation plant.

In 2006, a black liner was placed on the top of the BCR for two main purposes. First, the liner was installed to prevent significant snowmelt from entering the BCR every spring from the nearby the road and hillside. A small surface drain was placed at the opposite corner of where the snowmelt and water enter the BCR. Since the objective of the BCR project was to study passive treatment, it was hoped that the water would drain from the top of the reactor into the stormwater pond adjacent to the BCR. Second, the cover was installed in conjunction with an USEPA ORD project to measure the temperature at different levels inside the BCR that started in 2005. Because the BCR is at roughly a 7800 foot elevation, the USEPA ORD project manager believed that the black liner would be heated by sunlight and the temperature of the BCR would increase more rapidly. Heavy sandbags secured the liner.

Shortly after placing the liner on the BCR, several events occurred. The liner raised the temperature of the upper levels of the BCR, but some of this increase was also attributed to the exothermic reaction created by the microbes in the BCR. At first, this latter input was probably minimal, but later the gases began lifting the liner in many areas. This process continued throughout the treatment season and has occurred every year. The drain was effective at removing the water, but it appeared to remove too much water and the hydraulic head on the BCR changed and affected flow from the effluent line. The drain was plugged and the flow issue has not occurred again.

In 2007, Region 8, ORD, and CDM discussed the future of the system. The original design of the passive treatment system at Luttrell used two BCRs treating approximately 3-5 gpm each. Since the modeled flow from the final capped repository was estimated to be 6-9 gpm, changes would have to be made to the repository leachate piping, and the new BCRs would have to be located to allow for gravity draining. The existing pilot system does not have an adequate grade to drain into the leachate pond for pumping to the land application disposal site if the BCR is used without the precipitation plant. The metal loading to the land application disposal site would also have to be recalculated. The 2005-2006 experiments showed that mixing the BCR effluent with the leachate pond raised the leachate pH by 2 – 2.5 units, and it was decided to continue that approach. The BCR has been used for pre-treatment during 2007-2012, and CDM Smith has collected data during this time.

The BCR is the oldest constructed reactor at this elevation in the United States. The BCR is now 10 years old, and the last analyses completed by ORD in the fall of 2010 (Butler 2010) showed significant metals removal (aluminum, arsenic, cadmium, copper, and zinc are greater than 98%). In that same ORD peer-reviewed publication (Butler 2010), Luttrell had greater metal removal efficiency (MRE) for most metals, than the newer BCRs at Park City, UT and Crested Butte, CO.

B.15.5 MIW Water Chemistry and BCR Effluent Chemistry

The Luttrell BCR was built in September 2002 and the BCR was filled with repository leachate water over a period of weeks, so the influent water includes a range of values (see Table B.15-1). A substantial number of water chemistry analyses have been completed over the last 10 years, and it is difficult to summarize these in one short document. The influent water chemistry varies due to several factors and is highly depends on the amount of precipitation that enters the repository during the treatment season. During that time, a temporary cover may not have been over part of the repository, so the metals content of the leachate water would depend on percolation through the uncovered material in the repository. The uncovered material may have come from yard removal material (high in arsenic) or from material piles near abandoned mine sites in the superfund site boundaries.

The effluent from the BCR also varies with the time of year that it is collected. For example, the first effluent released when the BCR is started in the spring is black, with high biological oxygen demand (BOD), low dissolved oxygen (DO), low metals, and the lowest sulfate numbers for the year. This water has had a residence time in the BCR of at least 6 months, whereas effluent samples taken in September or October may have much shorter residence times (from a few days to a week or so). Table B.15-1 presents example data over eight years of the 10-year period. An upcoming publication will detail the water chemistry.

Table B.15-1. Luttrell site water chemistry summary (all data are mg/l; BDL= below detection level; NR= not reported)

Analyte Name	Initial Water Input to Cell 9/2002*	Initial Effluent Totals 04/29/03	Effluent 09/26/03	Effluent 8/18/04	Influent 7/2/2008	Effluent7/2/2008	Influent 9/30/2010	Effluent 9/30/2010
Aluminum	8 to 22	0.002	0.32	0.51	18.22	0.02	28	BDL
Arsenic	0.25	0.251	NM	NM	0.29	0.06	2.5	0.04
Calcium	340 to 354	45	1439.77	692.68	302	612	NR	NR
Cadmium	3.1 to 3.3	0.034	0.01	0.01	0.91	BDL	1.6	BDL
Cobalt	1.26 to 1.4	0.003	0.29	0.00	0.51	0.01	NR	NR
Chromium	0.032 to 0.253	BDL	0.02	0.01	0.01	BDL	NR	NR
Copper	4.16 to 5.2	0.395	0.02	0.01	33	0.01	12	0.07
Iron	1.6 to 1.64	10.9	9.19	2.72	1.38	3.29	27	0.28
Manganese	259 to 420	4.5	135.34	78.14	88.87	172.6	100	139
Nickel	0.71 to 0.76	0.034	0.20	0.03	0.31	0.01	0.31	BDL
Lead	0.04 to 0.05	0.086	BDL	BDL	0.1	0.01	BDL	BDL
Sulfur	833 to 849	41	985	732	515.56	584.44	NR	NR
Sulfate (ICP)	3,567 to 3,700	NM	2,955	2,196	NM	NM	4623	1315

Analyte Name	Initial Water Input to Cell 9/2002*	Initial Effluent Totals 04/29/03	Effluent 09/26/03	Effluent 8/18/04	Influent 7/2/2008	Effluent7/2/2008	Influent 9/30/2010	Effluent 9/30/2010
Zinc	36.7 to 330	3.39	24.01	0.52	126.78	0.38	270	0.18
pH					3.76	6.53		
Acidity	1.4	NM						
Total Alka-linity	BDL	NM			BDL	1170		

B.15.6 BCR Performance

Table B.15-1 shows the examples of influent and effluent concentrations over eight years. In general, the metal removal efficiencies for arsenic, aluminum, cadmium, cobalt, chromium, copper, nickel, lead, and zinc were 98% or greater. Manganese was high for the influent and even higher for the effluent.

June 2005 – Sept. 2005
Influent Sulfate Range 1350-5200 mg/l Effluent Sulfate Range 130- 1950 mg/l

For sulfate, at low influent concentrations, the effluent data show about a 75% sulfate concentration decrease. Sulfate reduction drops to about 30-33% at higher (> 4,000 mg/l) influent sulfate concentrations. Sulfate is available at this level, but it is possibly not distributed throughout the BCR well due to influent design. In addition, during the treatment season, the influent overloads the cell, which indicates that the residence time is insufficient. When the BCR is started in the spring, the BCR effluent sulfate concentrations are often below 200 mg/l after 6 months without new influent water.

June 2005 – Sept. 2005
Influent pH range 3.0 – 5.24 Effluent pH range 5.8 – 6.4

In 2005, as the influent sulfate increased, the pH decreased. This effect did not appear in the effluent. The effluent pH remained around a 6 pH range throughout the sampling season. The effluent pH was generally more than 2 units higher than the influent, which is expected with the bicarbonate alkalinity added by the microbial action and dissolution of limestone in the BCR.

Conductivity data are inconclusive. The field ORP measurements showed substantial results. The influent range was 217 – 417 mV, all positive readings as expected. The effluent ranged from -259 to -340 mV, with the negative range showing strong reduction throughout the sampling season.

B.15.7 BCR Monitoring

USEPA ORD monitored this BCR for almost 10 years, as did CDM Smith for Region 8. Complete data for both leachate (BCR influent) and BCR effluent are available for most years, and an upcoming publication will highlight the 10 years of operation. In addition, certain parameters are monitored on a daily basis with the operation of the chemical precipitation plant, especially when the BCR effluent is mixed with the leachate pond to increase the pH. A small chemistry lab exists on-site for this monitoring.

B.15.8 Regulatory Challenges

As with all CERCLA sites, differences have occurred among government entities working on the site. State and USEPA project managers have communicated with one another to manage these differences.

B.15.9 Stakeholder Challenges

The watershed has many stakeholders, including landowners, local communities, local and state government, special interest groups and several federal agencies, (including USEPA). Collaboration among the stakeholders is necessary to achieve a cleanup that is expedient, efficient, and long lasting. The Upper Tenmile site is being cleaned up using a collaborative, watershed approach. To date, USEPA has been unable to identify a potentially responsible party, so the cleanup on private land is paid with 90% federal funds and 10% state funds. Cooperating agencies have combined resources to expedite a watershed cleanup. The U.S. Forest Service has taken the lead role in cleaning up wastes on federal property within the Superfund site boundary (Beatrice, Justice, and Armstrong mines). Where individual mines involve both federal and private lands (Upper Valley Forge mine), cleanup expenses are shared by USEPA and the Forest Service, with the state of Montana paying 10%. The Luttrell repository was constructed as a regional repository to receive wastes from the Tenmile Creek and Basin Creek watershed sites. USEPA and the Forest Service share construction and maintenance costs of Luttrell repository.

Throughout the cleanup, USEPA has continued to work closely with the Forest Service, state, and local community. Community involvement is a priority in the Superfund process. USEPA uses a number of different tools and resources to promote effective, ongoing, meaningful community involvement. The goals of the Superfund community involvement program are as follows:

- Keep communities affected by sites informed throughout the cleanup process.
- Provide opportunities for communities to comment and offer their input about site cleanup plans.
- Facilitate the resolution of community issues tied to a site.

Throughout USEPA's Superfund response action at the Upper Tenmile Creek Mining Area site, the agency has provided information to the public through public meetings, the administrative record file for the site, fact sheets, and announcements published in local newspapers. USEPA

participates in monthly meetings of the Upper Tenmile Watershed Committee and routinely discusses the cleanup plans with area residents and property owners. In addition, during the field season, USEPA distributes periodic construction updates and maintains a project office in Rimini, where residents are welcome to stop by with questions. Tours and briefings are offered to local officials. USEPA also provides funding to Rimini Community, Inc., to obtain independent technical assistance though a Technical Assistance Grant (TAG). The TAG was awarded in 2005 and renewed through 2011. USEPA's Community Involvement Plan for the site is available at the following web page: http://www.epa.gov/region8/superfund/mt/upper_ten/CIP_0708.pdf.

B.15.10 References

Butler, B.A., M.E. Smith, D.J. Reisman, and J.M. Lazorchak. "Metal Removal Efficiency and Ecotoxicological Assessment of Field-Scale Passive Treatment Biochemical Reactors." *Environmental Toxicology and Chemistry* 30:1-8. 2010.

APPENDIX C. BCR CALCULATION EXAMPLES

C.1 Example 4-1, BCR Design Calculations

At a hypothetical site, a 7 m^2, 1-m deep BCR pilot test on mine water exiting a pilot-scale anoxic limestone drain (ALD) at 3.785 L/min has shown average sulfate reduction from 600 mg/L to 450 mg/L. The maximum total flow is 378.5 L/min and the copper concentration entering the BCR averages 1 mg/L. Although copper is the regulatory driver, the water also contains Zn at 2 mg/L.

The metal load that must be treated is calculated as mmol/d and compared to the volumetric rate of sulfide production in mmol/d/m^3:

$$(\sum C_{metal})Q = r_{SO_4\ reduction}$$

Where the volumetric rate of sulfate reduction, $r_{SO_4\ reduction} r_{SO_4\ reduction}$, is:

$$r_{SO_4\,reduction} = \frac{\Delta C_{SO_4} Q}{V}$$

The metal concentrations in moles are:

$$\text{Copper: } \frac{1\ ^{mg}/_{L}}{63.5\ ^{mmol}/_{mg}} = 0.0157\ ^{mmol}/_{L}\ \frac{1\ ^{mg}/_{L}}{63.5\ ^{mmol}/_{mg}}$$

$$\text{Copper: } \frac{1\ ^{mg}/_{L}}{63.5\ ^{mmol}/_{mg}} = 0.0157\ ^{mmol}/_{L}\ \frac{1\ ^{mg}/_{L}}{63.5\ ^{mmol}/_{mg}} \text{Copper: } \frac{1\ ^{mg}/_{L}}{63.5\ ^{mmol}/_{mg}} = 0.0157\ ^{mmol}/_{L}$$

$$\text{Copper: } \frac{1\ ^{mg}/_{L}}{63.5\ ^{mmol}/_{mg}} = 0.0157\ ^{mmol}/_{L} \text{Zinc: } \frac{2\ ^{mg}/_{L}}{65.4\ ^{mmol}/_{mg}} = 0.0306\ ^{mmol}/_{L}\ \frac{2\ ^{mg}/_{L}}{65.4\ ^{mmol}/_{mg}}$$

$$\text{Zinc: } \frac{2\ ^{mg}/_{L}}{65.4\ ^{mmol}/_{mg}} = 0.0306\ ^{mmol}/_{L}\ \frac{2\ ^{mg}/_{L}}{65.4\ ^{mmol}/_{mg}} \text{Zinc: } \frac{2\ ^{mg}/_{L}}{65.4\ ^{mmol}/_{mg}} = 0.0306\ ^{mmol}/_{L}$$

$$\text{Zinc: } \frac{2\ ^{mg}/_{L}}{65.4\ ^{mmol}/_{mg}} = 0.0306\ ^{mmol}/_{L}$$

and the moles of metal requiring treatment per day (with vain hope for 100% removal) are:

$$(\sum C_{metal})Q = (0.157 + 0.036)\ ^{mmol}/_{L}\left(378.5\ ^{L}/_{min}\right) = 17.5\ ^{mmol}/_{min} = 25.2\ mol/d$$

This means that the BCR should create not less than 25.2 mol S^{2-} per day. This design includes some conservatism because some of that sulfide will be lost to other fates such as evaporation of H_2S, unreacted outflow, and reaction with other cations. The rate of sulfide production will also decline with time as the substrate ages. The current volumetric reduction rate, assuming all sulfate reduction was biological, was:

$$r_{SO_4\,reduction} = \frac{\Delta C_{SO_4} Q}{V} = \frac{\frac{(600-450)\,mg/L}{96.06\,mg/mmol}\,3.785\,L/min}{V} = \frac{5.91\,mmol/min}{V}$$
$$= \frac{8.511\,mmol/d}{V} = \frac{8.511\,mmol/d}{V}$$

The volume of the pilot-scale BCR was 49 m^3 (= 7 m x7 m x 1 m), so the volumetric rate is:

$$\frac{8511\,mmol/d}{49\,m^3} = 174\,mmol/m^3 \cdot d$$

This rate falls nicely in the literature-reported range (100-300 mmol m^{-3} d^{-1}), so the design team is pleased with the pilot-scale success. A much higher rate is possible, but the designer should be concerned whether that rate is due to a short-term availability of highly degradable organic. A lower rate is also possible, particularly if the MIW has high levels of toxic metals such as copper or zinc. To account for other losses and aging, the designer arbitrarily assumes only half the produced sulfide will react with metal, so will use a sulfide production rate of only 87 mmol m^{-3} d^{-1} when calculating the required BCR volume.

The total volume of the BCR media is thus designed as:

$$\frac{(\Sigma C_{metal})Q}{r_{SO_4\,reduction}} = V$$

$$\frac{25.2\,mol/d}{0.087\,mol/m^3/d} = 290\,m^3$$

To not deviate from the pilot-scale unit, this volume will be laid 1 m deep. Two equal-size units will be designed so that one may be offline for maintenance. Each unit therefore has a footprint of 145 m^2.

C.2 Example 4-2, Hydraulic Calculations for a BCR.

The BCR system designed for the DJR site in Example 4-1 is a pair of downflow cells containing 1 m deep media with a footprint of 145 m^2 each. Each cell will resemble Figure C-1, with edges sloped at 1:2.

The site allows for ample area, so the design team chooses to have a shared wall (berm) and design each cell as a rectangle of 2L:1W. The interior footprint is 145 m^2, so the width is 8.5 m and the length is 17 m. The footprint of the top of the media is 10.5 x 21 m due to the 1:2 sloped sides.

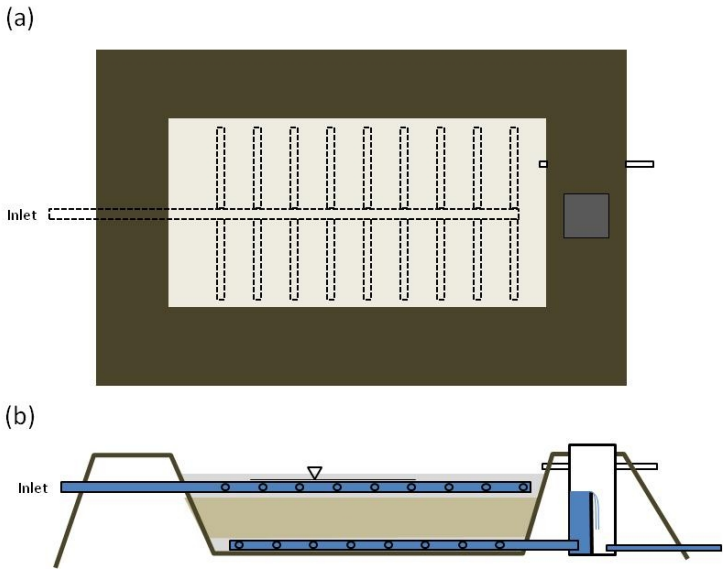


Figure C-1. BCR (a) plan view and (b) section view for Example 4-2, NTS.

Above and beneath the media there will be a 0.25-m (each) layer of gravel for water distribution and collection, and in the gravel will be a header pipe for delivery of water to perpendicular perforated pipes. These pipes can be thought of as dividing the bed into ten equal-length strips of 2.1 m (= 21 m/10). The maximum total flow is 378.5 L/min, but the average flow is 100 L/min per cell. Each cell therefore will operate at an average flow of 50 L/min, but might be tasked during a high flow event with half the maximum total flow, about 200 L/min.

The designer wishes to determine the estimated head loss through the bed at various flow rates. This example will; however, only illustrate these calculations at a flow of 100 L/min, the estimated maximum flow rate. Similar calculations would be done for the minimum and the maximum flow rates to determine how the changing flow will affect head/water level.

Piping:

The distribution piping is a header pipe (manifold) feeding the perpendicular pipes, which have holes to allow the flow to be fairly evenly distributed onto the bed, as shown in Figure C-2.

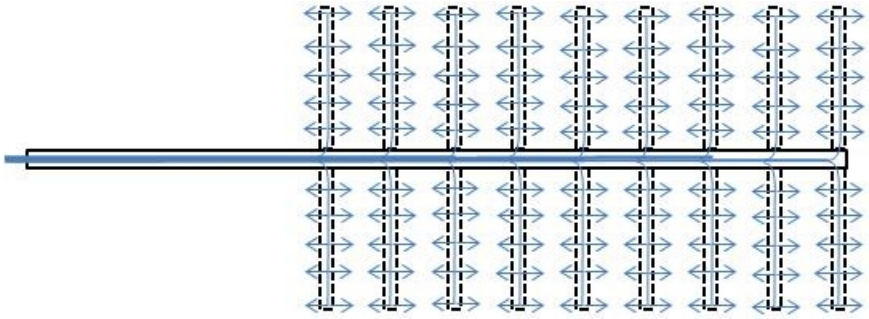


Figure C-2. The piping has two major components, in and out. The inlet is illustrated above. A larger header pipe serves as a manifold, connecting to the perpendicular perforated pipes, which allow water flow through the perforations. Water then flows through the gravel bed in which the pipes are set to the substrate. Water flows through the 1 m deep substrate bed, and then to a collection system, which mirrors the inlet system.

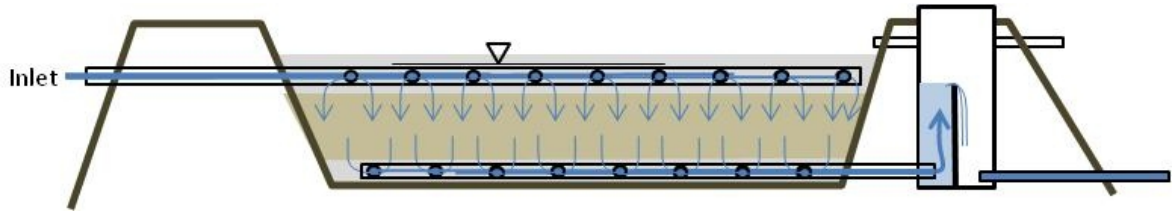


Figure C-3. Section view of influent-effluent control systems

The inlet header is at least 25 m (see Figure C-3; for reference bed length at top is 21 m) and the perpendicular distribution pipes are each 4.5 m long. The perforated pipes are located at 3.2 m (4 ft) on center. To minimize deposition, the designer considers a small enough pipe to reach 1 m/s at maximum flow:

$$v = 1\ ms^{-1} = \frac{Q}{A} = \frac{0.1\ m^3 s^{-1}}{\pi r^2} \rightarrow r = \sqrt{\frac{1}{\pi}\left(\frac{0.1\ m^3 min^{-1}}{(1\ ms^{-1})}\right)\frac{1\ min}{60\ sec}} = 0.023\ m$$

$$v = 1\ ms^{-1} = \frac{Q}{A} = \frac{0.1\ m^3 s^{-1}}{\pi r^2} \rightarrow r = \sqrt{\frac{1}{\pi}\left(\frac{0.1\ m^3 min^{-1}}{(1\ ms^{-1})}\right)\frac{1\ min}{60\ sec}} = 0.023\ m$$

The designer realizes that such a 1-inch pipe would clog easily, and makes a note to check on TSS in the MIW and if there is a lot of solids expected, to assure that pretreatment includes a sedimentation step. The designer then chooses a 6-inch PVC pipe as the 25-m long inlet header so it will be large relative to the flow rate, and minimize head loss. The head loss at average flow will be, using Darcy-Weisbach:

$$h = f\frac{L}{D}\frac{v^2}{2g} h = f\frac{L}{D}\frac{v^2}{2g}$$

where:

$$v = \frac{Q}{A} = \frac{0.1\ m^3\ min^{-1}}{\pi\left(\frac{0.1524\ m}{2}\right)^2} = 5.48\ {}^{m}/_{min} = 0.0914\ {}^{m}/_{sec}$$

$$\frac{Q}{A} = \frac{0.1\ m^3\ min^{-1}}{\pi\left(\frac{0.1524\ m}{2}\right)^2} = 5.48\ {}^{m}/_{min} = 0.0914\ {}^{m}/_{sec},$$

where:

L = length of pipe, 25 m,

D = pipe diameter, 0.15 m,

g = gravitational acceleration, 9.8 m s^{-2}, and

f = friction factor, found in this case from a Moody diagram to be 0.03.

$$\left(Re = \frac{Dv\rho}{\mu} = \frac{0.15 \times 0.0914 \times 999.7}{1.308 \times 10^{-3}} = 200\right)\left(Re = \frac{Dv\rho}{\mu} = \frac{0.15 \times 0.0914 \times 999.7}{1.308 \times 10^{-3}} = 200\right)$$

The head loss is therefore estimated as:

$$h = f\frac{L}{D}\frac{v^2}{2g} = 0.03\frac{25\ m}{0.15\ m}\frac{0.0914^2\frac{m^2}{s^2}}{2 \times 9.8\frac{m}{s^2}} = 0.002\ m$$

A very low head loss is expected for this large pipe carrying low-velocity water. However, the above calculation assumes all the water travels the full length of the pipe, but really the designer expects that some water will go into each perforated pipe, so the flow will decline along the length of this pipe. A designer could assume the 2 mm of head loss is a good enough estimate, expecting some other head loss will be much larger. However, this designer chooses to be more detailed in determining the head loss in the following spreadsheet. To evaluate the change in flow, one must assume the flow lost to each perforated pipe. An easy assumption is to divide by 18, assuming the

flow (100/18 = 5.56 L/min) goes equally into each perforated pipe. Thus the flow in the initial length of the pipe is 100 L/min, with 5.56 L/min going into the first perpendicular pipe and another 5.56 L/min in the paired pipe on the other side. Thus if the pipe is 8.2 m long until the pair of perforated pipes occurs, the flow at 9 meters past those pipes is 88.9 L/ min (= 100 – 2 x 5.56). The result is the spreadsheet shown in the table below.

$$A = -\frac{LQ}{K\Delta h} = -\frac{1\,m \times \left(0.00167\,{}^{m^3}/_{sec}\right)}{(3\times10^{-6}\,{}^{m}/_{sec})\times(-1\,m)} = 557\ m^2 A = -\frac{LQ}{K\Delta h} = -\frac{1\,m \times \left(0.00167\,{}^{m^3}/_{sec}\right)}{(3\times10^{-6}\,{}^{m}/_{sec})\times(-1\,m)} = 557\ m^2$$

Table C-1. Head loss calculation table

Remaining pairs perforated pipe	Length	Flow in pipe	Flow to perf. Pipe	Water velocity	Re	Friction factor from chart	h in section	h cumulative
	(m)	(L/min)	(L/min)	(m/s)			(mm)	(mm)
9	8.2	100	11.1	0.0914	1.0E+04	0.031	0.722	0.722
8	2.1	88.9	11.1	0.0812	9.3E+03	0.032	0.151	0.873
7	2.1	77.8	11.1	0.0711	8.1E+03	0.033	0.119	0.992
6	2.1	66.7	11.1	0.0609	7.0E+03	0.034	0.090	1.082
5	2.1	55.6	11.1	0.0508	5.8E+03	0.036	0.066	1.148
4	2.1	44.4	11.1	0.0406	4.7E+03	0.038	0.045	1.193
3	2.1	33.3	11.1	0.0305	3.5E+03	0.042	0.028	1.221
2	2.1	22.2	11.1	0.0203	2.3E+03	0.048	0.014	1.235
1	2.1	11.1	11.1	0.0102	1.2E+03	0.060	0.004	1.239

The above table shows that the head loss is less than if calculated for the whole pipe. As a percentage, the difference in values is large, but as a real value, 1.2 mm is very close to 2 mm.

The MIW flows to the perforated pipes at right angles to the header, with some head loss due to the joint. Calculating manifold head losses is potentially challenging but may be approached as assuming the velocity to be that in the distribution pipe and approximated as head loss to an orifice:

$$Q = \frac{\pi D^2}{4}\sqrt{2gh} Q = \frac{\pi D^2}{4}\sqrt{2gh}$$

However, the distribution pipe size is needed to solve this equation.

The designer decides that 3-inch (7.62 cm) drainage (perforated) pipe will be used for the 4.5-m long distribution piping. Smaller pipe would likely give acceptable head losses, but is not generally available. With the perforated pipe size known to be 3 inch, the head loss through the manifold's orifices can be determined:

$$Q = C\frac{\pi D^2}{4}\sqrt{2gh}$$

where:

Q = 0.0055 m^3/min (1/18th of flow),
D = 0.0762 m (3-inch diameter perforated pipe),
C = 0.82 because this orifice is really a square-edged entrance into a tube, and thus

$$\frac{1}{2g}\left(\frac{4Q}{\pi D^2}\right)^2 = h = \frac{1}{2 \times 9.8}\left(\frac{4 \times {}^{0.0055}/_{60}}{0.82 \times \pi \times 0.0762^2}\right)^2 = 0.0010 \text{ m}$$

Now the designer calculates the head loss down the perforated pipes. Using a similar process as shown above and continuing to assume the flow will be approximately equally distributed between the 18 lengths of pipe (Figure 4-5), the head loss estimated along the 4.5 m length is

$$v = \frac{Q}{A} = \frac{\frac{0.1}{18} m^3 \, min^{-1}}{\pi\left(\frac{0.0762 \text{ m}}{2}\right)^2} = 1.154 \, {}^{m}/_{min} = 0.01923 \, {}^{m}/_{sec}$$

$$\frac{Q}{A} = \frac{\frac{0.1}{18} m^3 \, min^{-1}}{\pi\left(\frac{0.0762 \text{ m}}{2}\right)^2} = 1.154 \, {}^{m}/_{min} = 0.01923 \, {}^{m}/_{sec} ,$$

$$h = f\frac{L}{D}\frac{v^2}{2g} = 0.06\frac{4.5 \text{ m}}{0.0762 \text{ m}}\frac{0.01923^2 \frac{m^2}{s^2}}{2 \times 9.8 \frac{m}{s^2}} = 0.000067 \text{ m}$$

Head loss along the perforated pipe is higher than calculated above due to the effective roughness due to the holes, but the effect is a few percent for low Re (Clemo 2006) and the head loss here is quite small. Further, as water is lost out the perforations, the flow, and thus velocity, will decrease. The head loss could be calculated with that flow decrease accounted for using a spreadsheet process as shown above. However, the error in estimating the head loss as 0.07 mm is trivial compared to other head losses, so the effort is not justified.

Next the water flows out of the perforations. The flow through each orifice (perforation) is described as a function of head loss:

$$Q = C\frac{\pi D^2}{4}\sqrt{2gh}Q = C\frac{\pi D^2}{4}\sqrt{2gh}$$

Assuming the perforated pipe follows a specification of having two rows of ¼-inch diameter holes at 4-inch spacing (center-to-center), then there are approximately 90 holes along the 4.5 m (15 ft) of pipe. Thus, each hole has a flow of 5.56 L/min/90 = 0.062 L/min. The flow per hole, Q, is thus 1.03 x 10^{-6} m^3/s, and the orifice equation can be solved for head loss to:

$$\frac{1}{2g}\left(\frac{4Q}{C\pi D^2}\right)^2 = h = \frac{1}{2 \times 9.8}\left(\frac{4 \times 1.03 \times 10^{-6}}{0.61 \times \pi \times 0.00635^2}\right)^2 = 0.000145 \text{ m}$$

where:

D = 0.00635 m (¼-in perforation), and

C = 0.61 because this orifice is a sharp-edged hole.

The pipe head losses at the inlet may be thus thought of as:

1. Flowing along inlet, loss of 0.7 – 1.2 mm,
2. Right turn into perforated pipe, loss of 0.1 mm,
3. Flowing along perforated pipe, loss of 0.07 mm, and
4. Flow through perforation, loss of 0.1 mm.

The total head loss depends on distance from the inlet, and will be in the range 1.0 – 1.5 mm.

If the water then goes into a gravel bed with the water level above the pipes, the head loss into that water layer will be the same from inlet to the top of the water. That means the head loss through any path from inlet to the water surface must be the same. The above analysis could therefore be iteratively solved by calculating the head loss at each perforation and varying the flow at each individual perforated hole (all 1000+) until the calculated head loss at each hole was equal. Such an approach was performed for this case and the calculated head loss was found to be 1.6 mm. At this point, the difference of perhaps 0.5 mm is likely to be insignificant, as the head loss through the bed should be very large.

The collection system pipes will have roughly the same head losses exiting the BCR because the same thing is happening but in reverse. The orifice losses might be a little different, using different coefficients, because of that reversed direction, but overall the expectation is for the same head loss overall. The outlet structure will need to be considered, and is separately considered a bit later in this example.

Gravel:

Water flows through the gravel between the piping and the substrate bed. The head loss due to the flow will be assumed to follow Darcy's law,

$$Q = -KA\frac{\Delta h}{L} Q = -KA\frac{\Delta h}{L}$$

where:

Q = flow through the gravel, which will be approximated as the total flow into the bed surface,

K = 10 cm/sec for well sorted gravel,

A = area of flow, which along with the path length, L, is interesting to determine.

The water flows out of a perforation in a pipe and into the gravel. The distance to flow (L) could be as little as straight down to the bed surface directly below the perforation, or could be as far as midway to a hole that is across and diagonal. Each perforation thus can be thought of as having a zone of influence that is rectangular and is illustrated below by orange rectangles (Figure C-4).

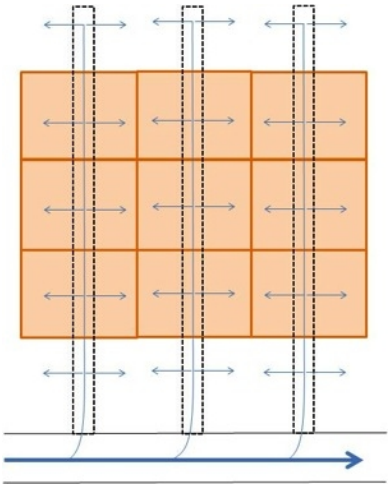


Figure C-4. Influent flow schematic

The majority of the rectangles are of dimension equal to the on-center spacing of the perforated pipes, 2.1 m and the spacing between the perforations, 4 inches (10.2cm). The area of flow is thus 0.0214 m^2. The distance to flow would best be represented as the geometric mean distance between the perforation and the edges of the rectangle. This rectangle (Figure 4-7), has an average distance from the perforation of about 52.5 cm, shown by the dotted blue line. The water also will sweep down through the gravel by the distance the perforated pipe is installed above the bed surface, which is assumed to have been specified as 6 inches (15.24 cm). This distance comes from the fact that the gravel bed was 15 inches deep, and if centered vertically the 3-inch perforated pipes would be 6 inches above the bed and 6 inches below the gravel surface. Factoring in that depth, and that water will not flow out 52 cm and then down 15 cm, but rather will flow fairly directly, the average path length should be 55 cm (Figure C-5). This value was calculated by considering the 52.5 cm path to be one arm of a 15.24 cm deep right triangle, with the hypotenuse describing the path length.

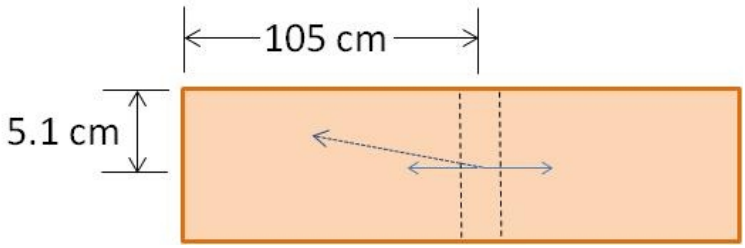


Figure C-5. Rectangle representing the zone of influence of the influent flow calculations from a perforated pipe

L is thus estimated as 55 cm. The area with that path length is the whole rectangle, 0.0214 m^2. The flow through that area was previously estimated as that from the pair of perforated holes, 2 x 1.03 x 10^{-6} m^3/s = 2.06 x 10^{-6} m^3/s, and the head loss is:

$$\Delta h = -\frac{LQ}{KA} = \frac{0.55\ m \times \left(2.06 \times 10^{-6} m^3/_{sec}\right)}{0.1\ ^{m}/_{sec} \times 0.0214\ m^2} = -0.00053\ m$$

$$\Delta h = -\frac{LQ}{KA} = \frac{0.55\ m \times \left(2.06 \times 10^{-6} m^3/_{sec}\right)}{0.1\ ^{m}/_{sec} \times 0.0214\ m^2} = -0.00053\ m$$

A small value such as this, 0.5 mm of head loss (note the negative sign, head was lost moving from the entry to exit on average, seems reasonable because gravel is easy to flow through.

The net head loss to get to the top of the substrate is about 2 mm.

Substrate:

The substrate bed is not easy to flow through, as the hydraulic conductivity (K) of such substrate is in the range 10^{-4} – 10^{-3} cm/sec is at least ten thousand times more difficult to flow through than is the gravel. To determine the head loss, Darcy's law is assumed to apply. The length to flow is 1 m, the depth of the bed. For this substrate the hydraulic conductivity was measured during the bench scale testing (Section 3.4) as 3 x 10^{-4} cm/s.

$$\Delta h = -\frac{LQ}{KA} = \frac{1\ m \times \left(Q m^3/_{sec}\right)}{\left(3 \times 10^{-6} m/_{sec}\right) \times A\ m^2} = \Delta h = -\frac{LQ}{KA} = \frac{1\ m \times \left(Q m^3/_{sec}\right)}{\left(3 \times 10^{-6} m/_{sec}\right) \times A\ m^2} =$$

The area of flow (A) is not trivial to determine. This area changes due to the sloped sides, which have a run of 1 m for every 0.5 rise. For the top of the substrate bed, the area is 10.5 x 21 m and at the bottom, 8.5 x 17 m. All of the flow goes through all of the bed, but the pathway along the sloped edge is longer than at the center where flow is likely straight down. Note the section view of the bed far above, and how one flow direction arrow curls along the sloped edge, illustrating this idea (Figure C-6). The edge is also a place to anticipate short circuiting, but one cannot pre-determine the amount of short circuit which will occur. The flow at the edge is suggested in the illustration below.

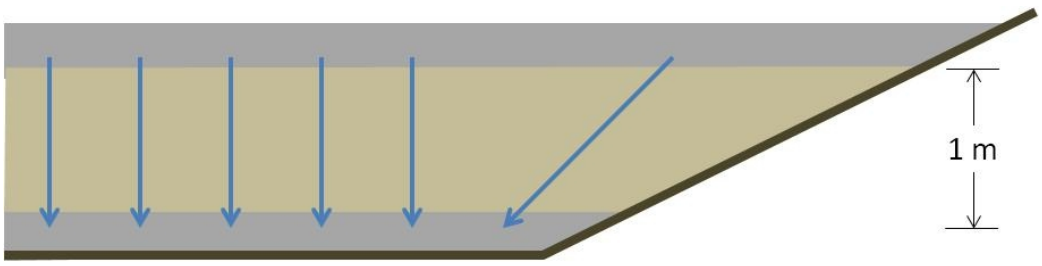


Figure C-6. Schematic showing the area of flow in a BCR containing sloped walls.

One potential significant error that could occur during construction is worth noting at this point: If the unit is lined, then the liner runs up the slope. The standard procedure to protect a liner is to cover it with a few inches of aggregate or sand. If a layer of sand or gravel runs up the slope, then this acts as a conduit for water flow (see Figure C-7).

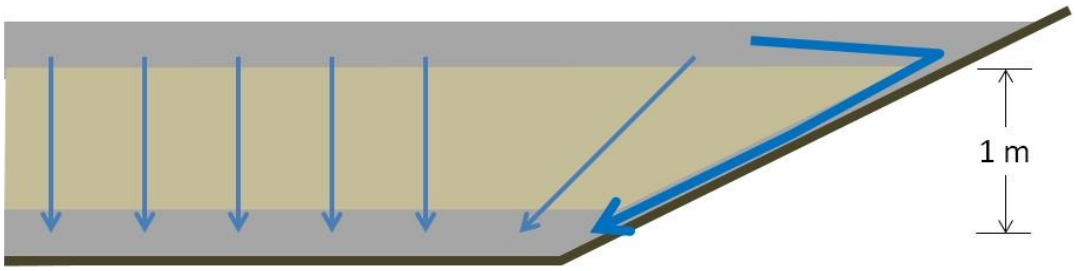


Figure C-7. Schematic showing the flow path as a result of using a gravel or sand bed on top of a liner in a BCR.

The image above might seem unreasonable in terms of the large flow through the thin aggregate (gravel) layer at the edge. However, that layer is thirty thousand times easier to flow through than is the same area of substrate (10 cm/sec compared to 3 x 10^{-4} cm/sec). If the aggregate was 4 inches (10 cm) thick, then the exposed edge is only a strip of 10 cm along each edge, with a total area to flow of (0.1 m x (2 x 10.5 m + 2 x 21 m) =) 6.3 m^2. The substrate bed area at the same surface is (10.5 m x 21 m =) 220.5 m^2, 35 times greater. However, that's 35 times more area that is 30,000 times more difficult to flow through. Comparing the value KA, the substrate is 0.0006615 m^3/s, while the gravel is 65 m^3/s. It would be a surprise if as much as 1% of the flow actually goes through the substrate.

Returning to the calculation of area, the flow all must go through the bottom of the bed, which is 8.5 x 17 m, or 144.5 m^2, which seems reasonable to use as A. The estimated head loss is:

$$\Delta h = -\frac{LQ}{KA} = \frac{1\ m \times \left(0.00167\ m^3/_{sec}\right)}{\left(3 \times 10^{-6} m/_{sec}\right) \times 144.5\ m^2} = -3.8\ m \Delta h = -\frac{LQ}{KA} = \frac{1\ m \times \left(0.00167\ m^3/_{sec}\right)}{\left(3 \times 10^{-6} m/_{sec}\right) \times 144.5\ m^2} = -3.8\ m$$

Thus the few mm of head loss in the distribution system is trivial when compared with head loss across the substrate! During average flow of 50 L/min, the head loss will be half as much, 1.9 m.

The designer will be challenged as to how to allow for this very large head loss, even the 1.9 m at average flow. If the water level is supposed to remain below the gravel surface, 15 cm above the substrate, then the amount of elevation change across the bed is about 1.3 m, and the discharge will effectively need to be another 0.6 m below the bottom of the bed.

One might think the discharge could be open to air. This would be problematic for two reasons. First, with no water trap on the line, air would get back into the bottom of the bed. Second, the

head loss is required for flow through the bed, so the water will need to be 1.9 m above the air, thus backing up 0.6 m above the gravel surface.

Most likely a standpipe of some sort will be installed to keep the bed from draining accidentally. Such a standpipe would be approximately level with the top of the substrate bed. In this case, the water level is expected to be 1.9 m above the bed, 1.75 m above the gravel layer. The sidewall depth would need to be at least 3.2 m (the sum of layer thicknesses plus the water depth) without allowing for freeboard. If the design allows for the full 100 L/min, then the water depth would be allowed to be 3.8 m and the sidewall would be at least 5.1 m. At 2:1 slope, this is a very large (about 30 m x 40 m) one-third empty pond (water footprint on average 21 m x 30 m).

Most likely, the designer will decide to make a larger bed with a resultantly lower head loss and more reasonable water depth. One might, for example, assume that a standpipe will be set at a level equal to the substrate surface, and decide the water level above that bed will be 1 m. Therefore the head loss across the whole bed would be 1 m. Assuming the substrate still dominates that flow, Darcy's law now would be solved for area:

If the length remains twice the width, $A = LW = 2WW$ solves to a width of about 17 m and length of 33.5 m, roughly four times larger than the original design.

APPENDIX D. WEIRS AND FLUME SIZE AND FLOW CALCULATIONS

D.1 Weirs

Selection of a weir should take into account the range of flow measurement. Recommended ranges are:

- high flows (132,000 gpm - 30,000 m^3/h: rectangular weir with or without final contractions.
- medium flows (50,000 gpm - 12,000 m^3/h): rectangular weir with or without final contractions.
- low flows (13,000 gpm / 3,000 m^3/h): triangular weir

Flow can be calculated using the following continuity equation:

$$Q = v\,A = C_d\,A\,(2\,g\,H)^{1/2}$$

where:

Q = Flow in units of volume/time; for example in m^3/s,

V = Fluid speed in units of length/time; for example in m/s,

A = Area where the fluid pass through with "V" velocity in units of surface; for example in m^2,

g = Gravitational acceleration constant: 9.81 m/s^2,

H = Water level in the weir, measured vertically from the head above the weir crest to the point of the effective head.

C_d = Discharge coefficient, determined experimentally

The following equations could be used in steady-state flow conditions with a free surface (discharge exposed to air) and employing dimensions recommended flows.

In order to obtain original equations, which can be used in a wide range of situations, please consult (Grant and Dawson 1997).

D.1.1 Broad-crested weir

$$Q\ [L/s] = 1{,}838\,L\,H^{1.5}$$

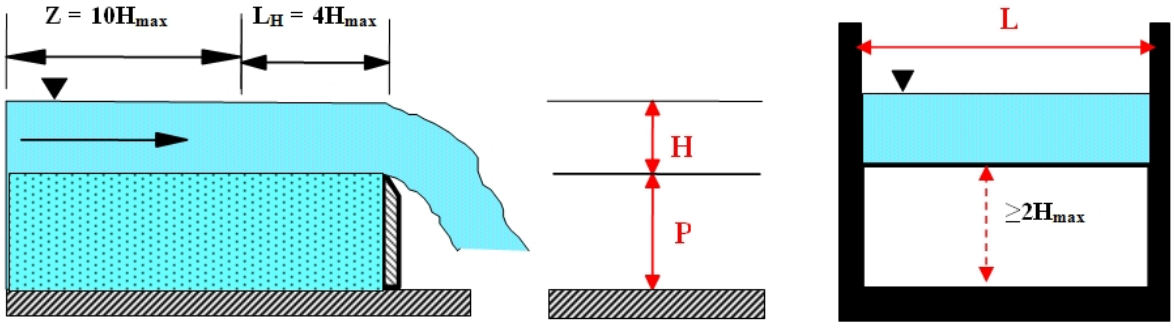
where:

Q = Flow (L/s)

H = Head of water approaching the weir (m)

L = Weir Length (m)

L_H= Measurement point of H (m)
Z = Proximity zone



$P \geq 0.3$ m; $H/P \leq 2.5$

Figure D-1. Rectangular weir without final contractions [H_{max} = maximum dimension = 2 * H = 2hmax].

Source: CPTS (2009).

Table D-1. Heights, minimal and maximal flows recommended for a rectangular weir without final contractions

Source: Grant and Dawson 1997

Length, L [m]	Minimum height H [m]	Maximum height H [m]	Minimum flow [L/s]	Maximum flow [L/s]
0.3	0.06	0.15	8.11	32.0
0.4	0.06	0.20	10.8	65.8
0.5	0.06	0.25	13.5	115
0.6	0.06	0.30	16.2	181
0.8	0.06	0.40	21.6	372
1.0	0.06	0.50	27.0	650
1.5	0.06	0.75	40.5	1,790
2.0	0.06	1.00	54.0	3,680
3.0	0.06	1.50	81.1	10,100

D.1.2 Rectangular weir

$$Q\ [L/s] = 1{,}838\ (L - 0.2\ H)\ H^{1.5}$$

where:
Q = Flow (L/s)
H = Head of water approaching the weir (m)

L = Weir Length (m)
LH = Measurement point of H (m)
Z = Proximity zone

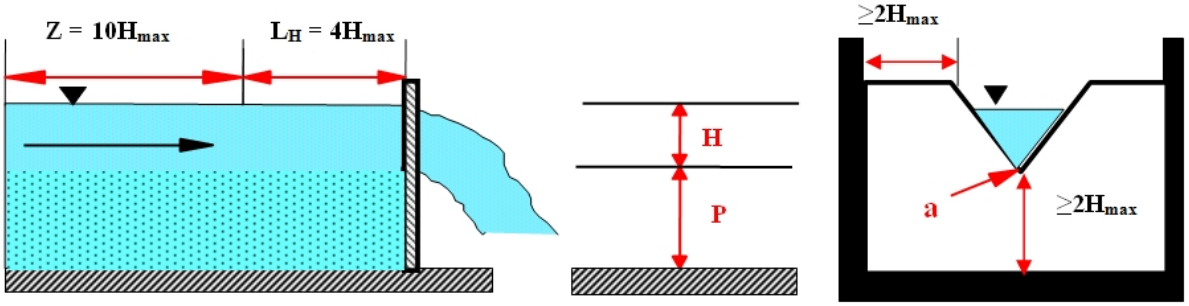


Figure D-2 Rectangular weir with final contractions [H_{max} = maximum dimension = 2 * H = 2hmax].

Source: CPTS 2009

Table D-2. Heights, minimal, and maximal flows recommended for a rectangular weir with final contractions

Source: Grant and Dawson (ISCO 1997)

Length, L [m]	Minimum height H [m]	Maximum height H [m]	Minimum flow [L/s]	Maximum flow [L/s]
0.3	0.06	0.15	7.8	28.8
0.4	0.06	0.20	10.5	59.2
0.5	0.06	0.25	13.2	103
0.6	0.06	0.30	15.9	163
0.8	0.06	0.40	21.3	335
1.0	0.06	0.50	26.7	585
1.5	0.06	0.75	40.2	1,610
2.0	0.06	1.00	53.7	3,310
3.0	0.06	1.50	80.7	9,120

D.1.3 Triangular weir (also called v-notch) (ISO 4371)

$$Q = \sqrt{\frac{g}{2}}\left(\frac{H}{0.795}\right)\tan\frac{a}{2}$$

where:

Q = Flow (L/s)

g = Acceleration due to gravity = 9.8066 m/s2

H = measured water depth and end of channel

a = Apex Angle: angle at the bottom of Triangle

Table D-3. Formulas to establish heights, minimal and maximal flows recommended for tri-angular weirs with different angles. Source: Grant & Dawson (ISCO 1997)

Angle (a)	Flow formula(1) [L/s]	Minimum height H [m]	Maximum height H [m]	Minimum flow [L/s]	Maximum flow [L/s]
30°	$Q = 373.2\ H^{2.5}$	0.06	0.6	0.329	104
45°	$Q = 571.4\ H^{2.5}$	0.06	0.6	0.504	159
60°	$Q = 796.7\ H^{2.5}$	0.06	0.6	0.703	222
90°	$Q = 1{,}380\ H^{2.5}$	0.06	0.6	1.220	385
120°	$Q = 2{,}391\ H^{2.5}$	0.06	0.6	2.110	667

(1) H, level in meters

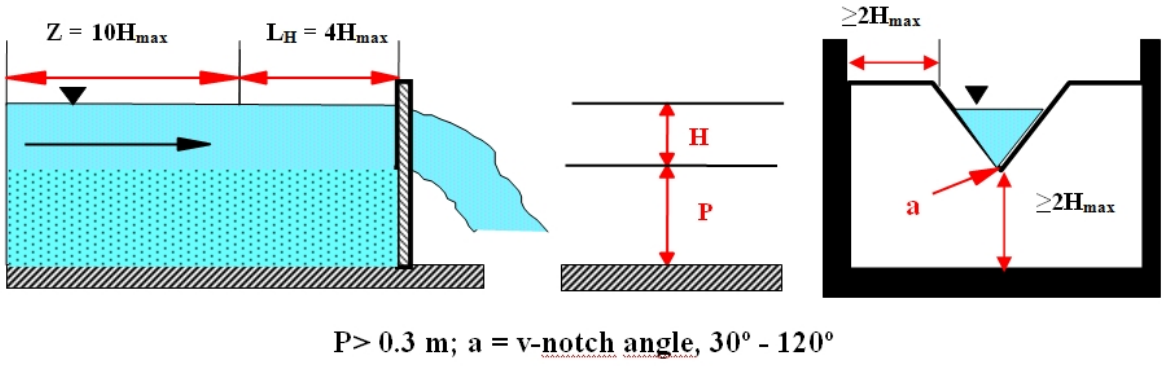


Figure D-3. Triangular weir [H_{max} = maximum dimension = 2 * H = 2hmax].

Source: CPTS 2009.

D.2 Parshall Flumes

The hydraulic equation to determine the flow using a Parshall flume is:

$$Q = x\ W\ (y\ H)^n$$

where:

Q = Flow, L/s

W = Throat width, m

H = Depth of a defined section before the obstruction, m

x, y, n = Calibration coefficients of the Parshall flume used

Table D-4 was elaborated using Parshall flume scheme shown in Figure D-4. According to AF-IPK/CNI (1999) the formula for the determination of the flow is:

$$Q\ [m^3/s] = 0.1132\ W\ (3.28\ H)^n$$

where:

Q = Flow, L/s

W = Throat width, m

H = Depth of a defined section before the obstruction, m

n = Constant

For W > 1 ft => n = 1.522 x W $^{0.026}$

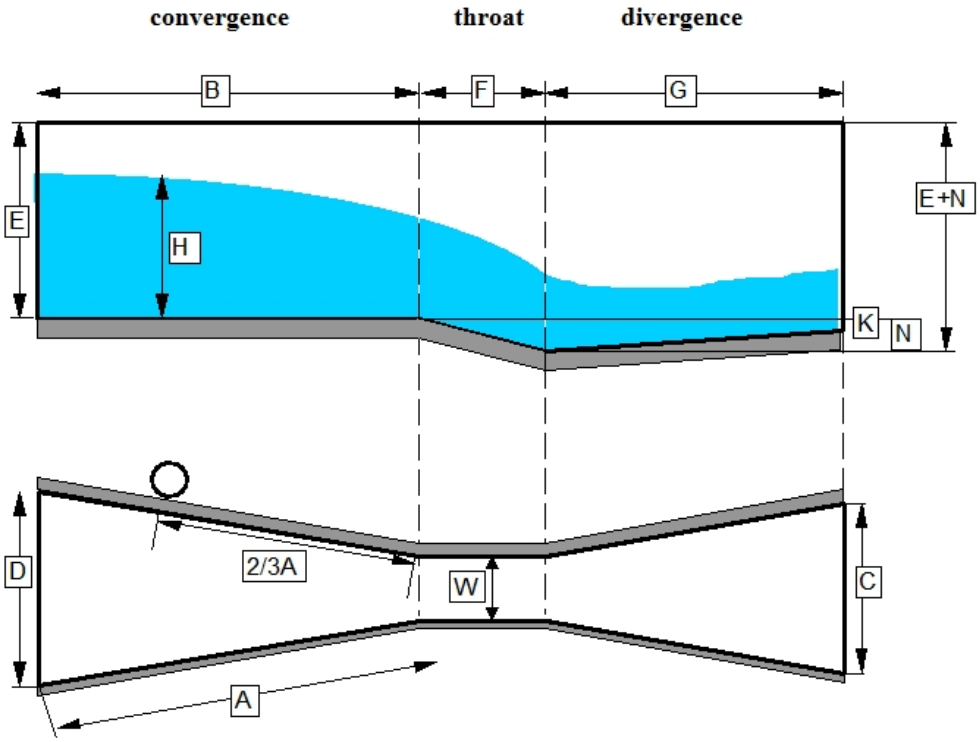


Figure 1. Parshall flume schematic.

Source: Fuente, AF-IPK/CNI (1999).

Table D-4. Data for the construction of Parshall flumes and for flow determination using depth and throat width

Source: Adapted from AF-IPK (1999)

W	W	W	A	2/3 A	B	C	D	E	F	G	K	M	N	n	MIN	MAX
in	ft	mm	mm	mm	Mm	mm	mm	mm	mm	mm	mm	mm	mm	constant	m^3/h	m^3/h
1	1/12	25.4	363	242	356	93	167	229	76	203	19		29	1.550	0.5	16
2	1/6	50.8	414	276	406	135	214	254	114	254	22		43	1.550	1.1	31
3	1/4	76.2	467	311	457	178	259	457	152	305	25	305	57	1.547	2.8	189
6	1/2	152.4	621	414	610	394	397	610	305	610	76	305	114	1.580	5.5	400
9	3/4	228.6	879	587	864	381	575	762	305	457	76	305	114	1.530	9.3	900
12	1	304.8	1,372	914	1,343	610	845	914	610	914	76	381	229	1.522	12	1,645
	1 1/2	457.2	1,448	965	1,419	762	1,026	914	610	914	76	381	229	1.539	17	2,500
	2	609.6	1,524	1,016	1,495	914	1,207	914	610	914	76	381	229	1.549	44	3,370
	3	914.4	1,676	1,118	1,645	1,219	1,572	914	610	914	76	381	229	1.566	63	5,140
	4	1,219.0	1,829	1,219	1,794	1,524	1,937	914	610	914	76	457	229	1.578	130	6,920
	5	1,524.0	1,981	1,321	1,943	1,829	2,302	914	610	914	76	457	229	1.587	158	8,730

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APPENDIX F. ACRONYMS

ALD	anoxic limestone drain
AMD	acid mine drainage
APC	aerobic polishing cell
ARD	acid rock drainage
ASTM	American Society of Testing and Materials
ATSDR	Agency of Toxic Substances and Disease Registry
BOD	biological oxygen demand
CEC	cation exchange capacity
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act
CFR	Code of Federal Regulations
COD	chemical oxygen demand
CWA	Clean Water Act
DO	dissolved oxygen
DOC	dissolved organic carbon
EBCT	empty bed contact time
EGL	energy grade line
GCL	geosynthetic clay liner
GPS	global positioning system
HDPE	high density polyethylene
HGL	hydraulic grade line
HRT	hydraulic residence time
ITRC	Interstate Technology and Regulatory Council
LBOS	limestone buffered organic substrate
LLDPE	linear low density polyethylene
LWF	lightweight fill
MCL	maximum contaminant level
MIW	mining-influenced water
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollution Discharge Elimination System
NPL	National Priority List
OLC	open limestone channel
OLD	open limestone drain
OSHA	Occupational Safety and Health Administration
PIG	pipeline inspection gauge
PLC	programmable logic controller

POC point of compliance
POTW publicly owned treatment works
PVC polyvinylchloride
QA/QC quality assurance/quality control
RAPS reducing alkalinity producing systems
RCRA Resource Conservation and Recovery Act
RFI request for information
RTD residence time distribution
SAPS successive alkalinity producing system
SMC spent mushroom compost
SMCRA Surface Mining Control and Reclamation Act
SRB sulfate reducing bacteria
TCLP Toxicity Characteristic Leaching Procedure
THPO Tribal Historic Preservation Officer
TMDL total maximum daily load
TSS total suspended solids
USDA United States Department of Agriculture
USEPA United States Environmental Protection Agency
USGS United States Geological Survey
VFP vertical flow pond
WET whole effluent toxicity

APPENDIX G. GLOSSARY

The following glossary provides definitions of words as used in this guidance document. In some cases the definition provided is specific for this document, rather than being more inclusive of other uses of the same word.

A

acid mine drainage (AMD)
A low pH, metal-laden, sulfate-rich drainage originating from a mined area that occurs where sulfur or metal sulfides are exposed to atmospheric conditions. It forms under natural conditions from the oxidation of sulfide minerals and where the acidity exceeds the alkalinity. See also acid rock drainage.

adsorption
Non-covalent bonding of a chemical to a solid surface.

aerobic polishing cell (APC)
A shallow pond which allows for aeration and settling of particles, typically following a BCR.

amorphous
Having no crystalline form.

areal loading rate
The amount of something applied to an area per time.

B

biochemical oxygen demand (BOD)
The amount of oxygen used as electron donor (food) is degraded by organisms.

biochemical reactor (BCR)
An engineered treatment system that uses an organic substrate to drive microbial and chemical reactions to reduce concentration of metals, acidity, and sulfate in mining-influenced water.

bog
A wetland that accumulates peat.

C

cell
An individual unit in a treatment system.

cellulose
An unbranched polymer of glucose found as the primary structural unit for green plants.

cobble
A stone measuring between 6.4 cm (2.5 in) and 25.6 cm (10.1 in) in size.

colloid
A particle which will not settle. Colloidal particles are generally 1 – 1000 nm in size.

constructed wetland
A man-made treatment system using saturated soils or sediment beneath standing water to remove contamination. Constructed wetlands almost always treat waste water of some type and almost always contain wetland plants.

co-precipitation
When a chemical is precipitated due to inclusion in a solid made from a different chemical.

D

Darcy's law
An equation which relates flow through a porous material to the driving force and the permeability of that material.

dispersed alkaline substrate (DAS)
Media containing a fine alkalinity source, often limestone sand.

dissolved organic carbon (DOC)
The weight of organic carbon in solution in a stated volume of water. The amount of dissolved organic carbon is usually determined by filtering water through a 0.45-µm pore-diameter filter, acidifying to drive off inorganic carbon, and then measuring the carbon dioxide that results when the carbon remaining in solution is oxidized.

divalent
Having two available outer shell electrons, generally resulting in a +2 or -2 charge.

E

Eh

The redox potential is the tendency of a compound to gain an electron. This is most often measured as the voltage required to prevent electrons to transfer between the measured sample and a standard reference electrode. For Eh, that standard reference, defined as zero volts, is $H2 \rightarrow 2\ H+ + 2\ e-$ at a specified standard condition.

electron acceptor

The molecule which is reduced during metabolism. In aerobic metabolism, oxygen is the electron acceptor, accepting two electrons and two protons to form water.

electron donor

The molecule which is oxidized during metabolism. For example, one glucose molecule used as an electron donor can result, with the addition of six water molecules during metabolic reactions, in six carbon dioxide molecules, 24 protons (H+), and 24 electrons (e-).

energy grade line (EGL)

A line that represent the elevation of energy head (feet/meters) of water flowing in a pipe, conduit, or channel. The line is drawn above the hydraulic grade line (gradient) a distance equal to the velocity head of the water flowing at each section or point along the pipe of channel (Sacramento State Office of Water, http://www.owp.csus.edu/glossary/energy-grade-line.php).

eutrophication

Excessive growth of phototrophs such as algae in water bodies due to the addition of nutrients.

F

fen

Fens are peat-forming wetlands that receive nutrients from sources other than precipitation, usually from upslope sources through drainage from surrounding mineral soils and from groundwater movement. Fens differ from bogs because they are less acidic and have higher nutrient levels. They are therefore able to support a much more diverse plant and animal community. These systems are often covered by grasses, sedges, rushes, and wildflowers. Some fens are characterized by parallel ridges of vegetation separated by less productive hollows. The ridges of these patterned fens form perpendicular to the downslope direction of water movement. Over time, peat may build up and separate the fen from its groundwater supply. When this happens, the fen receives fewer nutrients and may become a bog. See EPA website: http://water.epa.gov/type/wetlands/fen.cfm

flux
The mass (mass flux) or volume (flux) moving through an area per time.

fugacity
The chemical potential of a gas. For most gases at pressures around atmospheric, fugacity is equal to the pressure of the gas.

fulvic acids
A complex mix of the products of organic degradation which is resistant to further degradation and which are extracted into a strongly basic solution but will not precipitate with acid addition.

G

grub
To remove plants by digging up the roots and/or stumps.

H

head
A specific measurement of water pressure above a geodetic datum. It is usually measured as a water surface elevation expressed in units of length.

hemicellulose
A branched polymer of sugars used as a structural unit in many plants, generally linked to lignin.

hoop stress
The circumferential stress (perpendicular both to the axis and to the radius of the object in both directions of every particle in the cylinder).

humic acids
A complex mix of the products of organic degradation which is resistant to further degradation and which are extracted into a strongly basic solution but will precipitate when hydrochloric acid is added to adjust the pH to 1.

hydraulic loading rate
The volume of water applied to a system per time.

hydraulic residence time
The mean length of time a fluid element is in the volume of interest, usually determined by dividing the flow rate into the liquid volume.

K

kinetics
The study of rates of reaction.

L

labile
Able to change; highly degradable.

ligand
A chemical which interacts with a metal to bind that metal into a complex.

lignin
A complex, non-homogenous plant-made polymer found in unit walls cross linked to hemicellulose. Lignin is aromatic, hydrophobic, and resistant to biodegradation by most organisms.

limestone buffered organic substrate (LBOS)
A solid media containing limestone. Term is often used to describe the substrate used in vertical flow ponds.

loading
Mass of something per time entering a volume (volumetric loading rate) or flowing into an area (areal loading rate).

M

mass balance
An accounting for the mass entering, leaving, accumulating, and reacting in a system, often in the form of an equation such as "In – Out + Generation – Consumption = Accumulation".

monomer
The repeating unit of a polymer.

O

open limestone channel(OLC)
A trench containing limestone and open to the air into which water is introduced to raise pH and add alkalinity, with sufficient water velocity to prevent the deposition of precipitates resulting from metal (Al, Fe) oxidation . Also called an open limestone drain.

open limestone drain (OLD)
See open limestone channel; acronym also may describe

organic matter
Strictly defined, compounds in which carbon is bonded to hydrogen. Generally describes decomposed biological residues and other organic compounds synthesized by organisms.

overburden
Geological term for the material above solid rock. Sometimes called "soil".

oxidation-reduction potential (ORP)
The redox potential is the tendency of a compound to gain an electron. This is most often measured as the voltage required to prevent electrons to transfer between the measured sample and a standard reference electrode. ORP differs from Eh in that a variety of standard references may be used (often silver/silver chloride) and the resulting value can be adjusted to Eh by correcting for the electrode difference (e.g. 0 mV measured with a saturated KCl Ag/AgCl reference electrode is an Eh of 197 mV).

oxyhydroxide
A compound containing an unspecified arrangement of oxygen and/or hydroxides.

P

permeable reactive barrier (PRB
An engineered treatment system that uses a low hydraulic conductivity media to draw groundwater into contact with reactive substrate in order to reduce the concentration of one or more contaminants. Application of PRBs is governed by various patents (e.g. U.S. Patents 5,362,394 and 5,514,279).

polysaccharide
A polymer of sugars.

R

reduced
In chemistry, having gained electrons. Often gaining electrons is accompanied with gaining protons (hydrogen). As an example, when O_2 reacts with H_2, the oxygen is reduced, forming H_2O.

reducing alkalinity producing systems (SAP)
An engineered treatment system that uses an organic substrate to drive microbial reactions to reduce the concentration of free oxygen, followed by a carbonate source to increase alkalinity in mining-influenced water. Also called a successive alkalinity producing system and also called a vertical flow pond.

reducing conditions
A system in which the gain of electrons is energetically favored due to a low reduction potential.

rhizosphere
The root zone of plants.

ruminant
An animal that digests plant matter in a process which allows microbes to break the cellulose and hemicellulose in the plant matter into digestible break-down products.

S

sedimentation
The process of depositing entrained particles from water.

semi-passive
Requiring some mechanical and/or energy to operate such as timed valves ort dosing pumps, but not major pieces of equipment.

sludge
A watery semi-solid.

solubility product (Ksp)
The equilibrium constant used to determine the amount of a chemical in solution. The value is related to the product of multiplying the molar concentrations of what dissolves, so the smaller the value, the less is the dissolved concentration.

stoichiometry
The relative quantities of molecules involved in a reaction.

substrate
Either (a) a chemical which reacts or (b) a solid surface or (c) an electron donor.

successive alkalinity producing systems (SAP)
An engineered treatment system that uses an organic substrate to drive microbial reactions to reduce the concentration of free oxygen, followed by a carbonate source to increase alkalinity in mining-influenced water. Also called a reducing alkalinity producing system and also called a vertical flow pond.

sulfate reducing bacteria (SRB)
Single-celled organisms in the bacteria domain which are able to use sulfate as an electron acceptor.

sulfate reducing bioreactor (SRBR)
An engineered treatment system that uses an organic substrate to create sulfate reducing conditions and drive microbial and chemical reactions to reduce concentration of metals, acidity, and sulfate in mining-influenced water.

sulfate reduction
The stripping of oxygen atoms from sulfate (SO_4^{2-}), most often yielding sulfide (S^{2-}) as an ultimate product.

supersaturated
Present in a nonequilibrium condition where the concentration is above the saturation limit.

suspended solids (SS)
The weight of material able to sediment out of a stated volume of water. The amount of suspended solids is usually determined by filtering water through a pre-weighed 0.45-µm pore-diameter filter, drying the filter, and determining the mass gained.

T

total dissolved solids(TDS)
The weight of matter, including both organic and inorganic matter, dissolved in a stated volume of water. Often used as a surrogate for the concentration of salts in solution. The amount of dissolved solids is usually determined by filtering water through a glass or 0.45-µm pore-diameter micrometer filter, weighing the filtrate residue remaining after the evaporation of the water, and drying to constant weight.

total suspended solids(TSS)
The weight of material able to sediment out of a stated volume of water. The amount of suspended solids is usually determined by filtering water through a pre-weighed 0.45-µm pore-diameter filter, drying the filter to constant weight, and determining the mass gained.

V

vertical flow pond (VFP)
An engineered treatment system that uses an organic substrate to drive microbial reactions to reduce the concentration of free oxygen, followed by a carbonate source to increase alkalinity in mining-influenced water. Also called a successive alkalinity producing system and also called a reducing alkalinity producing system.

volatile suspended solids (SS)
The weight of material able to sediment out of a stated volume of water which will, effectively, burn away. Generally used as an estimate of the mass of organic particulate matter such as biomass. The amount of volatile suspended solids is usually determined by filtering

water through a pre-weighed 0.45-µm pore-diameter filter, drying the filter, and determining the mass lost when the filter is incubated at 500 C or higher.

W

whole effluent toxicity (WET)

Aggregate toxic effect to aquatic organisms from all pollutants contained in a facilities wastewater (EPA, http://water.epa.gov/scitech/methods/cwa/wet/)