Passive Treatment of Mine Impacted Water In Cold Climates: A Review

YUKON RESEARCH CENTRE
Yukon College

This publication may be obtained from:

Yukon Research Centre, Yukon College
520 College Drive
P.O. Box 2799,
Whitehorse, Yukon
Y1A 5K4
(867) 668-8895
1-800-661-0504
www.yukoncollege.yk.ca/research

Recommended citation:

Ness, I., Janin, A. and Stewart, K. Passive Treatment of Mine Impacted Water In Cold Climates: A review. Yukon Research Centre, Yukon College.

Printed in Whitehorse, Yukon, 2014 by Arctic Star Printing Inc., 204 Strickland St.
Photo credit: André Sobolewski

Executive summary

In this review, we evaluate the challenges encountered and the adaptations required for the successful treatment of mine-impacted waters in cold climates with Passive Treatment Systems (PTSs). Engineered PTSs are modeled on natural wetlands, which have been shown to effectively treat water with high metal concentrations through natural attenuation. PTSs include constructed wetlands, bioreactors, and hybrid systems. Some of the challenges associated with implementing cold climate PTSs include cold temperatures, remote locations and limited access in winter, which can lead to freezing pipes and surface water, variable seasonal flow, and low productivity of microbial and macrophytic communities. Many adaptations have been implemented to address these cold climate challenges including burial of pipes to avoid hydraulic failure, insulation to avoid freezing surface waters, bypasses and overflows to maintain constant flow, summer establishment of microbial and macrophytic communities and the addition of liquid carbon sources to offset reduced organic matter decomposition in cold temperatures. While further investigation and development is necessary to fully understand the factors affecting cold climate PTSs, with sufficient research and planning PTSs can be successfully implemented in cold climates.

Acknowledgements

This review was funded by a NSERC Applied Research and Development grant held by Yukon College and Casino Mining Corporation. We would like to thank Andre Sobolewski of Clear Coast Consulting for his expertise and contribution to the document.

Table of contents

List of Figures

List of Tables

1. BACKGROUND

Passive treatment of mine impacted water has become increasingly common over the last 30 years. While natural attenuation of contaminants in wetlands, bogs and swamps has been well documented (McGregor, 2002; Sobolewski, 1997), recent advances in the design and construction of engineered passive treatment systems (PTSs) has resulted in more efficient contaminant removal (USEPA 2014; INAP 2010; Mays & Edwards, 2001; MEND 1996). PTSs consist of constructed wetlands (CWs), bioreactors, or some combination and/or variation on both of these components. Other types of PTSs exist, such as in-pit or in-pond treatment systems, but these are more specialized and will not be reviewed here.

Natural wetlands provide important information about the processes that drive PTSs, as well as, acting as water treatment systems in their own right. Natural wetlands and bogs have been removing contaminants from water for centuries (Gusek, 2008). From cupriferous and uraniferous bogs (Sobolewski, 2013) to bog iron ore deposits (Gusek, 2008), many examples of natural attenuation of metals in wetlands exist (e.g., Boyle, 1995; Wieder and Lang, 1982). Numerous studies have found evidence of metal retention in natural wetlands from natural and anthropogenic sources (Table 1). Natural wetlands can also give us an indication of the potential for the long-term performance of CWs. A wetland in Ireland has been assimilating metal-laden water from an abandoned Pb/Zn mine for over 120 years without interference (Sheoran and Sheoran, 2006). A natural wetland receiving drainage high in Fe, Zn and Mn from an iron ore mine in the Rocky Mountains was found to retain metals at a rate of 90%, 65% and 25%, respectively (August et al, 2002). The wetland received mine drainage for over a century and had accumulated large amounts of metal oxides and sulfides. This likely resulted in a decrease in efficiency overtime, as at the time of the study the wetland was found to be a sink for all metals during the winter, but then became a source for Zn and Mn during the summer when flows were higher. Similarly, a fen in Northern Manitoba treating Ni mine drainage was found to remove 96% of the Ni (Hambley, 1996). While higher levels of Ni were observed in the effluent in the spring and after heavy rains, sequential extractions of the soils indicated that 97% of the Ni captured by the fen would be unlikely to be remobilized. In another natural wetland receiving drainage from an abandoned Pb/Zn mine in Ireland, retention of 95% of Zn and 65% of As was observed (Beining and Otte, 1997). The wetland has been receiving mine drainage for over a century and is estimated to have reached only 30% of its capacity. A wetland receiving coal mining drainage high in Se removed 83% of the inflowing metal concentration (from 364 mg/L to 22mg/L) and selenium-reducing bacteria and SRB were found at high levels (Baldwin and Hodaly, 2003). Finally, a natural wetland established below the Silver King mine in Elsa, Yukon, was shown to completely remove Zn from mine drainage during winter months, even when temperatures decreased below -40°C (Sobolewski, 2003a). Evidence of sulfide generation was uncovered during this study, indicating that SRB were active and could account for metal removal.

Natural wetlands have been treating mine-impacted water in cold climates since mining activity began and while several studies indicate their efficacy and longevity, their effectiveness over long time periods and the effects of seasonal variation on efficiency still remains poorly documented. Understanding the effects of changes in water flow through the seasons would better inform PTS designers to adapt their own systems for cold climates. Natural wetlands also help to inform the scale necessary for treatment wetlands. The longevity of natural wetlands, particularly in cold climates, indicates that a well-designed PTS may last for decades to centuries.

Table 1: Natural wetlands shown to ameliorate mine drainage (Sobolewski, 1999, p.26).

Wetlands that only partly improve mine water quality					
Mine	**Location**	**Dominant plant Species**	**Acidic Drainage**	**Water quality parameter**	**Ref.**
Abandoned tin, zinc mines	St Hilary Mining District, West Cornwall, U. K	*Juncus effusus, Phragmites australis*	No	Cu yes, Zn no	Brown, 1997
Carbonate mine	Montana	*Carex rostrata*	Yes	Al no, Fe yes, Pb yes	Dollhopf *et al.*, 1988
Coal mines	Eastern United States	*Typha, Scirpus* spp.	Yes	pH, Al, Fe, Mn	Kleinmann, 1996
Coal mine	Mpumalanga mining district, South Africa	*Typha,* spp.	Yes	pH, Al, Fe, Mn	Limpitlaw and Bloem, 1996
Dunka Mine	Minnesota, United States	Peat bogs	Yes	Cu, Ni	Eger *et al.*, 1994
Equity silver	British Columbia, Canada	Sedges	Yes	Cu	Mehling, 1985
Mt. Washington Mine	British Columbia, Canada	*Eriophorum angustifolium*	Yes	Cu, Al	Coombes, 1998
Natural wetlands at 35 coal mines	Pennsylvania, USA	*Mainly Typha,* spp., others also noted	Yes	pH, Al, Fe, Mn	Stark, 1990
Ranger Mine	Jabiru, Australia	*Eleocharis sphacelata* *Typha* species	No	U	Jones *et al.*, 1996
St. Kevin Gulch	Colorado, USA		Yes	Fe yes, Zn no	Walton-Day, 1994
Unnamed coal mine	Ohio, USA	*Sphagnum* spp.	Yes	pH, Fe, Mn	Huntsman, Solch and Porter, 1978
Tug Fork coal mine	West Virginia, U.S.A.	*Sphagnum* spp.	Yes	pH, Fe, Mn	Wieder and Lang, 1982

Wetlands which improve mine water quality to full environmental compliance					
Mine	**Location**	**Dominant plant Species**	**Acidic Drainage**	**Water quality parameter**	**Ref.**
Abandoned Lead/ Zinc mines	Glendalough, County Wicklow, Ireland	*Molinia caerulea, Juncus acutiflorus*	Yes	As, Cd (?), Pb (?), Zn	Beining and Otte, 1996
Birchtree Mine	Manitoba, Canada	*Agrostis, Carex* spp. *Typha latifolia*	No	Ni	Hambley, 1996
Cluff Lake, Rabbit Lake	Saskatchewan, Canada	*Carex* species *Sphagnum* moss	No	U	TAEM, 1995; Sinclair, 1996
Con Mine	Northwest Territories, Canada	*Carex* species *Sphagnum* moss	No	CN^-, As	Ball, 1993
Hilton Mine	Mt. Isa, Queensland, Australia	Reeds, algae	No	Fe, Mn, Tl, Zn	Jones and Chapman, 1995
Pacific Mine	Utah, USA	?	No	Cd, Pb, Zn	Lidstone and Anderson, 1993
Quirke Mine	Ontario, Canada	*Typha latifolia*	Some	Fe, Ra-226	Davé, 1993
Silver Queen	British Columbia, Canada	Sedges	No	Zn	Higgs, 1996
Star Lake, Jolu	Saskatchewan, Canada	*Carex* species *Sphagnum* moss	No	CN^-, Cu	Gormely *et al.*, 1990
Tom's Gully Gold Mine	Darwin, Australia	*Melaleuca, Typha domingensis*	No	As, Co, Cu, Fe, Pb, Mn, Ni, Zn	Noller, Woods and Ross, 1994
United Keno Hill Mines	Yukon Territories, Canada	*Carex* species *Sphagnum* moss	No	Fe, Mn, Pb, Ag, Zn	Boyle, 1965
Woodcutter's Mine	Darwin, Australia	*Typha* species	No	Cd, Mn, Pb, Zn	Noller, Woods and Ross, 1994

In Yukon, Canada, effluent from hard rock mines can have neutral or acidic pH and be high in a suite of heavy metals, depending on the local geology (Nordin, 2010). Contaminated mine drainage often requires treatment prior to discharge to the receiving environment. Yukon mines have expressed interest in implementing PTSs to treat contaminated drainage and in some cases have already begun research to develop such systems (i.e. Minto Mine, Casino Mine, Faro Mine, United Keno Hill Mine, Brewery Creek, Eagle Gold and Wolverine Mine). There are distinct advantages to PTSs in remote locations with limited access and power. Passive systems should ideally be able to function for decades with no power (Gusek & Wildeman, 2002). While most systems are not truly "walk-away" scenarios, they are generally lower maintenance than active systems (Johnson & Hallberg, 2005).

Some of the challenges facing PTSs in cold climates include hydraulic failure due to freezing in winter, reduced microbial activity, and increased water flows during spring melt. Many adaptations have been developed to overcome these challenges such as insulating and/or burying PTSs, maintaining constant flows through the use of bypasses and overflows, and providing carbon sources to feed bacteria during times of slower organic matter decomposition. There have been varying degrees of success, but hurdles have been overcome and many advances have been made in the use of PTSs in cold climates.

With the proliferation of PTSs around the world and the growing interest in Yukon, it is necessary to synthesize some of the successes and failures of PTSs, particularly in a northern context. This review aims to summarize many of the PTSs that have been used to treat mine-impacted water and address some of the challenges and future research directions associated with implementing these systems in cold climates. This review focusses primarily on PTSs for the treatment of mine-impacted water, although systems that treat other types of effluent are discussed where relevant cold climate adaptations have been used.

2. **PASSIVE TREATMENT SYSTEMS**

PTSs are generally defined as water treatment systems that require little or no maintenance once installed (Johnson & Hallberg, 2005). This is contrasted with active treatment systems (treatment plants), which use mechanical devices operated by workers and require continual flow control, reagent input, sludge/residual removal and power. More specifically PTSs can be described as water treatment systems that sequentially remove contaminants using biological or geochemical processes (Gusek & Wildeman, 2002). They are typically designed to last for decades and be less costly over the long term than active treatment systems. However, they can also require much more space and may be constrained by site limitations, such as topography (Johnson & Hallberg, 2005).

PTSs capitalize on natural attenuation processes that naturally occur in wetlands. Wetlands generally have an aerobic zone where plants thrive and oxidative conditions dominate. Below this is an anaerobic zone where reductive processes dominate and sulfate-reducing bacteria (SRB) thrive (Gusek, 2002). In each zone, different reactions occur that precipitate different contaminants. In the aerobic zone, hydroxide and oxide precipitation occurs and in the anaerobic zone sulfide and carbonate precipitation is catalyzed by microorganisms. Some contaminants like chromium, selenium or uranium are retained by reductive immobilization in these anaerobic environments. Filtration and adsorption by detritus are also important processes occurring in both zones. PTSs promote these processes by using organic materials as substrates for microbial activity, as well as, local native plant species as the living wetland layer. The chosen substrate is often waste material that can be obtained at low cost.

Additionally, organic material such as wood chips or sawdust can often be obtained on-site or nearby, reducing or eliminating transport costs (Gusek, 2002).

Metal removal in PTSs can be divided into abiotic or biotic processes. Technologies that rely on abiotic processes include anoxic limestone drains (ALDs), open limestone channels, diversion drains, vertical flow limestone drains and others. All these systems rely on limestone to neutralize acid drainage and remove aluminum and iron. They are often used in combination with other biotic passive treatments. For example, successive alkalinity producing systems (SAPS) or vertical flow limestone drains rely on the percolation of acidic water through a layer of compost, where the carbon dioxide produced by its microbial decomposition greatly enhances limestone dissolution (Kepler & McCleary, 1994). These systems are used to treat waters with high acidity, aluminum or ferric iron, which cannot be treated by other limestone-based PTSs.

Systems that rely on biotic processes can be divided into aerobic and anaerobic systems. Aerobic wetlands rely on oxidative processes to remove metals (primarily iron) and are most often constructed to treat mine water that has net alkalinity (Hedin et. al., 1994, Johnson & Hallberg, 2005). Anaerobic wetlands, or bioreactors, are used to treat neutral, metal rich water by reductive processes. They are typically applied at base and precious metal mines to remove metals that were originally present as sulphides and were released during the mining process (Sobolewski, 1999). Constructed wetlands and bioreactors are further detailed in Section **Error! Reference source not found.** and **Error! Reference source not found.** below.

The technology behind PTSs has evolved considerably over the last 20 years. A landmark development was the publication of an Information Circular by the former US Bureau of Mines (Hedin et al, 1994). This document presented for the first time design criteria that were field-validated, along with a decision-making flow chart that identified the different types of treatment required for different types of mine waters. The information circular focused on acid drainage from abandoned coal mines in Eastern Appalachia, where the authors had gained their experience. More recently, Gusek (2008) expanded this flow chart by including additional forms of treatment that were predominantly developed for abandoned base and precious mines in Western USA and Canada. This chart presents the multitude of PTS components and possible configurations available for industry and those involved in reclamation (Figure 1).

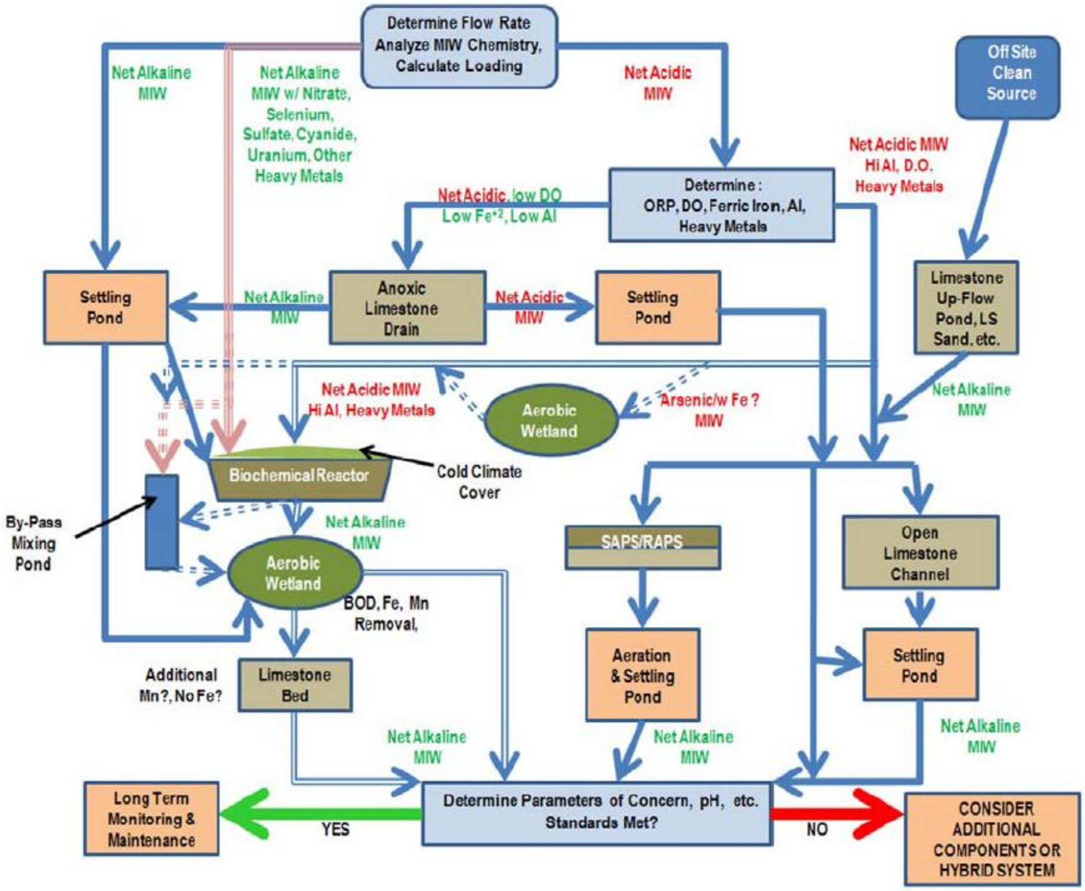

Figure 1: Flow chart for selecting PTS based on water chemistry and flow (adapted from Hedin et al, 1994 by Gusek, 2008, p. 4).

2.1. Constructed wetlands

Constructed wetlands (CWs) are defined as engineered wetlands that use natural processes associated with vegetation, soil and microbes to treat contaminated water (Mander & Jenssen, 2003). The basic processes that operate in CWs include nutrient uptake, bacterial degradation, filtration, oxidation, precipitation, sedimentation and adsorption (Sheoran & Sheoran, 2006). The characteristics of wetlands that contribute to their effectiveness in treating contaminated water include: high productivity, microbial diversity, organic matter accumulation, high reactive surface area and slow water flow (Sobolewski, 1999). It is these characteristics that designers seek to emulate when creating CWs, and many CWs are more efficient at removing metals than natural wetlands (Sheoran & Sheoran, 2006). CWs are used all over the world for treating various types of effluent. They have been used extensively in cold climates for municipal wastewater treatment (Mander & Jenssen, 2003) and in several northern settings for mine drainage (Nyquist & Greger, 2009; Sobolewski 1996a; Sobolewski 1996b; Duncan, 2002; Goulet & Pick, 2001).

Surface flow wetlands often consist of an excavated pond lined with an impermeable substrate, such as clay or geotextile and are filled with porous materials like peat (Eger & Eger, 1991), or sand (Nyquist & Greger, 2009), then fertilized (Sobolewski, 1996b) and planted with local wetland plants (Figure 2a). Sub-surface flow wetlands are similarly constructed but attempt to direct the flow through the root zone of the plants (Figure 2b). In both cases, the wetland bed is gently sloped so that water flows by gravity.

In Europe, sub-surface flow CWs are most often used to treat municipal wastewater (Galletti et al, 2010; Hiley, 2003; Vyzamal & Krasa, 2003), whereas in North America, they are most often used to treat mine drainage. Native wetland plants are generally chosen based on their ease of propagation and transplantation, water depth tolerance, root structure, and metal accumulation properties. Many studies have assessed the role of plants in CWs and most have found that plants uptake only a small fraction of the metals sequestered by a CW with most of the metal retention occurring in the roots (Noller et al, 1994; Sobolewski, 1999; August et al, 2002; Deng et al, 2004; Galletti et al, 2010; Mitsch & Wise, 1998; Nyquist & Greger, 2009; Taylor & Crowder, 1983; Vyzamal & Krasa, 2003; Ye et al, 2001) or on the surface of belowground biomass in a metal-rich plaque layer (Weis & Weis, 2004). It has been hypothesized that low translocation of metals from roots to shoots is a strategy of metal-tolerant plants (Taylor & Crowder, 1983; Deng et al, 2004). Metal uptake in plants varies considerably depending on the plant species, contaminants being treated, and the pH of water and sediment (Weis & Weis, 2004).

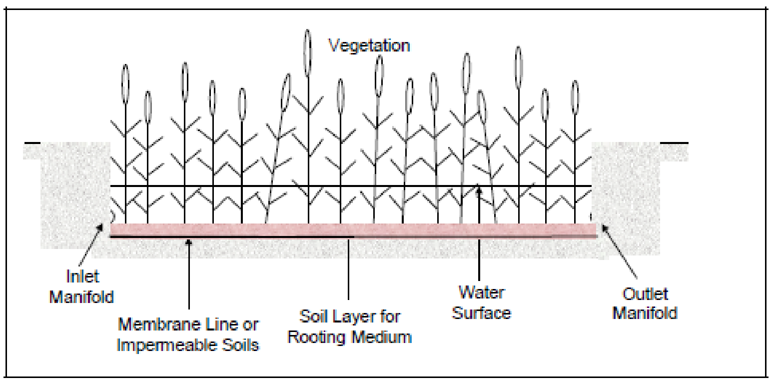

Figure 2a: Surface flow wetland (adapted from a drawing by SC Reed from EPA, 2000a, p. 1).

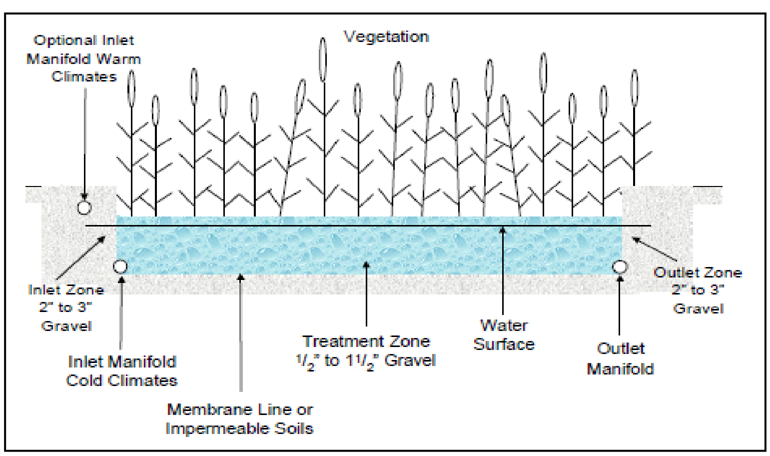

Figure 2b: Sub-surface flow wetland (adapted from a drawing by SC Reed from EPA, 2000b, p. 1).

2.2. Bioreactors

Bioreactors (or anaerobic wetlands) are PTSs that are contained in an excavation, pond, tank or even a mine shaft. It is filled with a solid reactive mixture made up of sand, gravel, compost, manure, wood chips, sawdust, or other organic material from nearby sources (Figure 3) (Schmidtova & Baldwin, 2011). The most effective reactive mixture generally contains an organic carbon source, a bacterial inoculum, a solid porous medium, a nitrogen source and a neutralizing agent (Zagury & Neculita, 2007). As mine water percolates vertically or horizontally through the substrate material, decomposing organic matter provides nutrients that fuel the activity of sulfate reducing bacteria (SRB) and other microorganisms that provide treatment (Hashim et al, 2011). The SRB then produce bicarbonates, which increases the pH while reducing sulphates to sulphides, which in turn precipitate with metals to form highly insoluble metal sulphides (Zagury & Neculita, 2007). Some bioreactors also rely on other anaerobic bacteria to remove nitrate (Reinsel, 1998) or selenium (Pahler et. al., 2007).

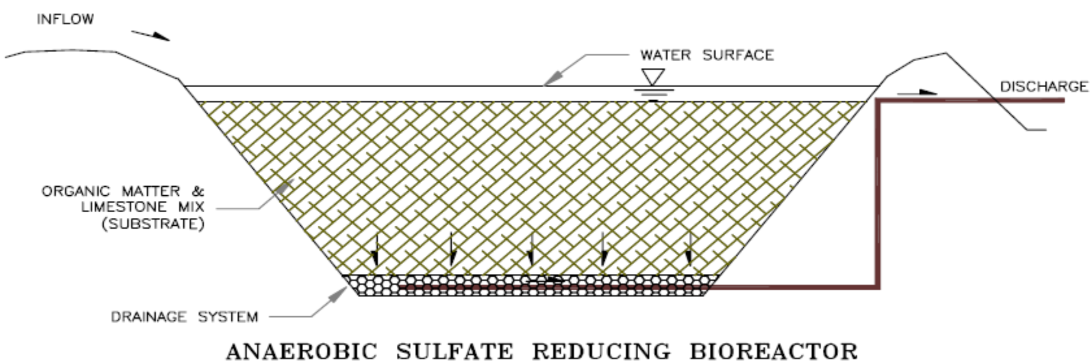

Figure 3: Typical bioreactor design (Gusek, 2002, p. 3).

Bioreactors can be used in combination with other PTS components such as limestone pretreatment to increase pH, constructed wetlands and aerobic cells (Figure 4). Bioreactors have been used at numerous mine sites in cold climates with limited access and power (Duncan et al, 2002; Ettner, 2007; Kuyucak, et al, 2006; Nordwick et al, 2007; Sobolewksi, 2010; Zagury & Neculita, 2007) due to low construction and operating costs (Schmidtova & Baldwin, 2011) and their ability to work year-round in sub-zero temperatures (Kuyucak et al, 2006).

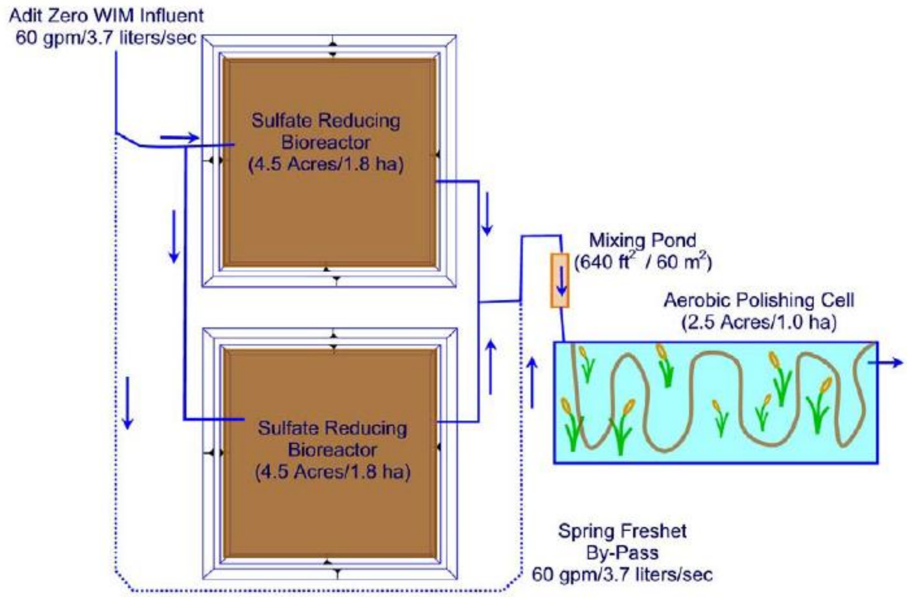

Figure 4: Hypothetical hybrid system with bioreactor and aerobic wetland components (Gusek & Conroy, 2007, p. 265).

2.3. Pilot and full scale PTSs

Many PTSs are currently being employed to treat contaminants in northern and/or cold climate environments (Table 2). While many lab and bench scale experiments are documented in the literature, our focus is only on those that have reached pilot and full scale implementation. For the purposes of this review, full scale passive treatment projects are defined as those that are constructed on-site to deal with a large portion of the effluent being produced. Pilot scale projects are experimental PTSs and are generally implemented on-site to increase the likelihood of successful full scale design (Gusek, 2002). An overview of pilot and full scale PTSs studies is provided in Table 2 including the metal removal rates observed. Note that this survey is not fully comprehensive and only reflective of the available literature, as there are PTSs that have operated for years, but have never been documented.

Table 2: Overview of Pilot and Full Scale Passive Treatment Systems

Passive Treatment Type	Components	Target Contamination	Location	Scale	Duration	Removal rates	Reference
Bioreactor	Bioreactor cell	Zn, Fe, Mn, Ni, Cd, Cu,	United Keno Hill Mine, Yukon, Canada	Pilot scale	2 years	Zn (98-99%), Mn (99%), Sb (80%), As (58-80%)	Alexco, 2012
CW	HSSF wetland & aerobic polishing cell	Cu, Zn, Pb, Fe	Clear Creek/Central City, Colorado, USA.	Pilot scale	n/a	Cu (99%), Zn (98%), Pb (94%), Fe (86%)	Banks et al, 1997
CW	Aerobic wetland and permeable reactive barrier	Fe, Mn, Al, Ni	Northumberland, UK	Full scale	2 years (ongoing)	Fe (99%), Mn (66%), Ni (83%), Al (80%)*	Batty & Younger, 2004
Bioreactor	BCR, aerobic polishing cell, aeration cell	Se, Tl, Zn	Montana, USA	Pilot scale	4 months (ongoing)	Se & Tl (99.5%), Zn (97%)	Blumenstein & Gusek, 2008
CW	3 different wetlands constructed in duplicate	Cu, Fe, S, Se	Minto Mine, Yukon, Canada	Pilot scale	1 year (ongoing)	Cu (89%), Se (26%)	Contango Stategies, 2013
Hybrid	ALDs, sedimentation ponds, CW, SAPS	Fe	Howe Bridge SAPS, Pennsylvania, USA	Full scale	7 years	Fe (50-69%)	Demchak et al, 2001
Bioreactor	SAPS, settling pond, SAPS	Fe, Al	Filson SAPS, Pennsylvania, USA	Full scale	4 years	Fe (90%), Al (20-50%)	Demchak et al, 2001
Bioreactor	Holding pond, SAPS	Fe, Al	Sommerville SAPS, Pennsylvania, USA	Full scale	3 years	Al (30%), Fe (increased)	Demchak et al, 2001
Bioreactor	SAPS, settling pond	Fe, Al	McKinley SAPS, Pennsylvania, USA	Full scale	2 years	Fe (92%), Al (25-90%)	Demchak et al, 2001
Hybrid	2 vertical upflow anaerobic bioreactors, 3 HSSF wetlands, sand filter, holding pond	As, Zn, Cd	Teck Metals Smelter, Trail, BC, Canada	Full scale	5 years (ongoing)	91% to 100% net removal efficiency	Duncan et al. 2002; Duncan, 2010
CW	5 surface flow wetland systems	Ni	Dunka Mine, Minnesota, USA	Small scale	18 years (ongoing)	Ni (30-90%)	Eger et al, 1991; Eger & Eger, 2005; Eger & Beatty, 2013
Bioreactor	Pretreatment pond, 2 bioreactors, settling pond, flushing pond & aeration channel	Al, Cu, Fe, Ni, Se, Zn	Leviathan Mine, California, USA	Full scale	2 years (ongoing)	Al (99%), Cu (99%), Fe (95%), Ni (86%), Zn (98%), Se (20%)	EPA, 2005
Bioreactor	Vertical flow cell, limestone drain, settling pond	Fe, Al, Mn	Indiana County, Pennsylvania, USA	Full scale	7 years	Al (Up to 90%)	Eppley & Gusek, 2009

Cold climate PTSs are highlighted in grey.

*calculated based on available published data. ALD=anoxic limestone drain, BCR=biochemical reactor, CW=constructed wetland, HSSF=horizontal subsurface flow wetland, SAPS=successive alkalinity producing system.

Table 2 (continued): Overview of Pilot and Full Scale Passive Treatment Systems

Passive Treatment Type	Components	Target Contamination	Location	Scale	Duration	Removal rates	Reference
Bioreactor	Limestone drain, 4 anaerobic basins	Cu, Zn, Cd, Pb, Cr, Ni	Kongens Mine, Norway	Pilot scale to full scale	4 years (ongoing)	Cu (85%), Zn (48%), Cd (98%), Pb (82%), Cr (71%), Ni (24%)	Ettner, 2007
Bioreactor	4 anaerobic basins	Ni. Al, Cu, Cd, Zn, Cr	Titania mine, Norway	Pilot scale to full scale	8 years (ongoing)	Ni (35-98%), Al (98%), Cu (98%), Cd (98%), Zn (99%), Cr (64%)	Ettner, 1999
CW	2 HSSF wetlands	Cu, Ni, Zn	Ferrara, Italy	Pilot scale	1 year	Cu (3.4-9%), Ni (25-35%), Zn (26%)	Galletti et al, 2010
CW	Surge pond, 4 sub-surface flow anaerobic cells, 2 aerobic polishing cells	Fe, Zn, Mn, Cd, Cu, Zn, As	Butte, Montana, USA	Pilot scale	2 years	Fe (37%), Zn (99%), Mn (45%), Cu (97%), Cd (99%), As (-38%)*	Gammons et al, 2000
CW	Liming circuit, free water ponds & permeable treatment walls	Fe, Zn, Mn, Cd, Cu, Zn, As	Butte, Montana, USA	Pilot scale	2 years	Fe (99%), Zn (99%), Mn (99%), Cu (97%), Cd (97%), As (53%)	Gammons et al, 2000
Bioreactor	Biofilter	As	Wood Cadillac Mine, Quebec, Canada	Full scale	7 years	As (81%)	Germain & Cyr, 2003; Tasse et al, 2003; Libeiro, 2007
CW	Surface flow wetland	Fe, Mn	Kanata, Ontario	Full scale	2 years (ongoing)	Fe (15-31%), Mn (23-49%)	Goulet & Pick, 2001
CW	CW with permeable barriers	Cu, Zn, Cd, Pb, Ni, Cb, Fe, Mn, As	Curilo, Bulgaria	Full scale	10 years	n/a	Groudev et al, 2008
Bioreactor	Settling basin, 2 anaerobic bioreactors, rock filter & aeration pond	Pb, Zn, Cd, Cu	West Fork, Missouri, USA	Full scale	4 years (est. 30 yr lifetime)	Pb (90%), Zn (80%), Cd (>33%), Cu (>78%)*	Gusek et al, 2000
Bioreactor	Vertical flow BCR, mixing basin, flow dispersion zone	Fe, Al, Cu, Zn, Cd, Mn	Golinsky Mine, California, USA	Full scale	6 years (ongoing)	Zn (~80%), Fe (~99%), Cu (~97%)*	Gusek et al, 2011

Cold climate PTSs are highlighted in grey.

*calculated based on available published data. ALD=anoxic limestone drain, BCR=biochemical reactor, CW=constructed wetland, HSSF=horizontal subsurface flow wetland, SAPS=successive alkalinity producing system.

Table 2 (continued): Overview of Pilot and Full Scale Passive Treatment Systems

Passive Treatment Type	Components	Target Contamination	Location	Scale	Duration	Removal rates	Reference
Hybrid	Vertical flow BCR, mixing pond, vegetated aerobic polishing cell	Fe, Cu, Zn, Cd	Iron King/Copper Chief Mine, Arizona, USA	Full scale	3 years (ongoing)	Al (50%), Cu (93%), Zn (96%), Cd (>89%), Mn (20%)*	Gusek et al, 2013
Hybrid	2 anaerobic bioreactors & aerobic surface flow wetland	Fe, Zn, Cu, Cd, Al	Haile Gold Mine, South Carolina, USA	Full scale	5 years (ongoing)	Fe (85%), Zn (80-100%), Cu (53%), Cd (94%), Al (97%)*	Gusek & Schneider, 2010
CW	6 connected ponds & CW	Fe	Marchand PTS, Pennsylvania, USA	Full scale	7 years (ongoing)	Fe (99%)	Hedin, 2013
CW	2 cell CW	Fe, Al	Quaking Houses, UK	Full scale	1 year (est. 15-20 yr life)	Fe (45%), Al (63%)	Jarvis & Younger, 1999
Bioreactor	Collection ditch, anaerobic cell, oxidation pond & limestone drain	Ni, Cu, Zn, Fe, Al, Mn	Cadillac Molybdenite Mine, N.Quebec, Canada	Full scale	2 years (ongoing)	Al (78%), Cu (97%), Fe (99%), Mn (40%), Ni (98%), Zn (99%)*	Kuyucak,2006
CW	HSSF wetland	Ni, Al, Cu, Pb, Zn, Cr, Fe, Mn	Flanders, Belgium	Full scale	3 years (ongoing)	Ni (49%), Al (93%), Cu (90%), Pb (53%), Zn (87%), Cr (66%), Fe (-50%), Mn (-103)	Lesage et al, 2007
CW	Surface flow wetland	Cr, Ni, Zn	Bahco metallurgical plant, Argentina	Full scale	2 years (ongoing)	Cr (86%), Ni (67%), Zn (95%)	Maine et al. 2006
CW	4 cell CW	Fe, Mn, As	Jackson County, Alabama, USA	Full scale	15 years	Fe (98%), Mn (79%), As (100%)	Mays & Edwards, 2001
CW	3 cell CW	Fe, Mn, Zn, Cd, B, As, Pb	Stevenson, Alabama, USA	Full scale	14 years	Fe (97%), Mn (47%), Zn (33%), Cd (100%), B (52%), As (99%), Pb (26%)	Mays & Edwards, 2001
CW	6 CW cells & 2 anaerobic cells	Fe, Al	Ohio, USA	Pilot scale	>1 year	Fe (84%), Al (36%)	Misch & Wise, 1998
Hybrid	Oxidation pond, surface flow wetlands, vertical flow bioreactors, re-aeration ponds, limestone beds & polishing pond	Fe, Zn, Pb, Cd, As, Ni	Mayer Ranch PTS, Oklahoma, USA	Full scale	4 years (ongoing)	Fe (99%), Zn (99%), Ni (96%)*	Nairn, 2013

Cold climate PTSs are highlighted in grey.

*calculated based on available published data. ALD=anoxic limestone drain, BCR=biochemical reactor, CW=constructed wetland, HSSF=horizontal subsurface flow wetland, SAPS=successive alkalinity producing system.

Table 2 (continued): Overview of Pilot and Full Scale Passive Treatment Systems

Passive Treatment Type	Components	Target Contamination	Location	Scale	Duration	Removal rates	Reference
Bioreactor	Subsurface bioreactor	Fe, Al, As, Zn, Mn, Cd, Cu	Lilly/Orphan Boy Mine, Montana, USA	Full scale	11 years	Fe (65%), Zn (99%), Al (>99%), Mn (75%), As (96%), Cd (>98%), Cu (>99%)*	Nordwick et al, 2006
Bioreactor	Pretreatment, above & below ground bioreactors	Fe, Al, Mn, As, Cd, Cu	Calliope Mine, Butte, Montana, USA	Full scale	3 years	Fe (99%), Al (99%), Mn (32%), As (0%), Cd (19%), Cu (98%)*	Nordwick et al, 2006
Bioreactor	Anaerobic reactor, passive alkalinity addition, anaerobic reactor, passive aeration, aerobic reactor	Fe, Zn, Mn, Al, As, Cd, Cu	Surething Mine, Montana, USA	Pilot scale	Ongoing	Mn (99%) & As (92%)	Nordwick et al, 2006
CW	4 surface flow wetlands	Fe, Zn, Cu, Cd	Kristineberg mine, Sweden	Pilot scale	1 year	Cu (36-57%)	Nyquist & Greger, 2009
CW	Horizontal flow wetland, cascade filters	Mn, Cu, Zn, Cd, Pb, Cr	Sewage Processing Plant, Przywidz, Poland	Full scale	2 years (ongoing)	Pb (60%), Cu (57%), Zn (50%), Cd (67%), Mn (75%)*	Obarska-Pempkowiak & Klimkowska, 1999
CW	Sedimentation pond, aerobic cells, anaerobic cells	Zn, Mo	Antamina Mine, Peru	Full scale	9 years (ongoing)	Zn (51%), Mo (65%)	Plewes et al, 2009
Bioreactor	Vertical flow BCR	Se	Grand Junction, Colorado, USA	Pilot scale	1 year	Se (98%)	Rutkowski et al, 2010
CW	2 CWs (1 large & 1 small)	Cu	Bell Copper Mine, Smither BC, Canada	Pilot scale	2 years	Cu (40-98%)	Sobolewski et al, 1994
CW	CW	Zn	United Keno Hill Mine, Yukon, Canada	Pilot scale	4 months	Zn (90%)	Sobolewski, 1996b
Bioreactor	5 anaerobic cells	Al, Fe, Cd, Cu, Zn	Tulsequah Chief Mine, BC, Canada	Pilot & Full scale	4 years	Al (92%), Fe (97%), Cd (46%), Cu (80%), Zn (32%)*	Sobolewski, 2010.
Bioreactor	2 parallel bioreactors	Se	Smoky River coal mine, BC, Canada	Pilot scale	1 year	Se (97%)	Sobolewski, 2010.
CW	Surface flow wetland	Cu	Campbell Mine, Ontario, Canada	Full scale	14 years	Cu (90-95%)	Sobolewski, pers comm.

Cold climate PTSs are highlighted in grey.

*calculated based on available published data. ALD=anoxic limestone drain, BCR=biochemical reactor, CW=constructed wetland, HSSF=horizontal subsurface flow wetland, SAPS=successive alkalinity producing system.

Table 2 (continued): Overview of Pilot and Full Scale Passive Treatment Systems

Passive Treatment Type	Components	Target Contamination	Location	Scale	Duration	Removal rates	Reference
Hybrid	Fixed film aerobic reactor, biological trickle filter	Al, Fe, Mn, Sr, Zn	Merrick Landfill, North Bay, Ontario, Canada	Pilot scale	1 year	Al (80%), Fe (96%), Mn (87%), Sr (41%), Zn (60%)*	Speer et al, 2012
Hybrid	Fixed film aerobic reactor, sand & gravel CW	Al, Fe, Mn, Sr, Zn	Merrick Landfill, North Bay, Ontario, Canada	Pilot scale	1 year	Al (80%), Fe (95%), Mn (60%), Sr (41%), Zn (60%)*	Speer et al, 2012
CW	Pretreatment basin, aeration cascade, limestone channel & 2 wetlands	Fe, Mn, Zn, As	Jales mining site, North Portugal	Full scale	6 years (ongoing)	n/a	Valente et al, 2012
CW	2 parallel sub-surface horizontal flow CWs	Mn, Al, Cu, Zn	Prague, Czech Republic	Full scale	3 years (ongoing)	Mn (80.9%), Al (>82.3%), Cu (>91.1%), Zn (>97.5%)	Vyzamal & Krasa, 2003
Bioreactor	5 BCRs	Fe, Al	Tab Simco coal mine, Illinois, USA	Pilot scale	4 months	Fe (56%), Al (95%)	Walters et al, 2013
CW	Vegetated wetland and stabilization pond	Pb, Zn, Cd	Pb/Zn mine, China	Full scale	16 years (ongoing)	Cd (94%), Pb (99%), Zn (97%)	Yang et al, 2006
CW	4 wetland cells	Fe, Mn, Co, Ni	Springdale, Pennsylvania, USA	Full scale	2 years (ongoing)	Fe (94%), Mn (98%), Co (96%), Ni (63%)	Ye et al, 2001

Cold climate PTSs are highlighted in grey.

*calculated based on available published data. ALD=anoxic limestone drain, BCR=biochemical reactor, CW=constructed wetland, HSSF=horizontal subsurface flow wetland, SAPS=successive alkalinity producing system.

3. NORTHERN ENVIRONMENTS

PTSs in northern environments are often located in remote locations with limited access and power, short growing seasons and cold winter temperatures, conditions which can present challenges for PTS design. These challenges however, are also often reasons for choosing PTSs, which can be constructed to operate year-round with little maintenance and no power.

For the purposes of this review, cold climates are described as those that may experience freezing conditions during the winter months. This definition allows the inclusion of any PTS that may experience seasonal variations in efficiency and may employ adaptations for cold temperatures. The main problems faced by PTSs in cold climates are hydraulic failure due to freezing, ice formation and insufficient biological activity at low temperatures (Wittgren & Maehlum, 1997; Mander & Jenssen, 2003). Due to the interest in implementing PTSs in cold climates and the need to develop site-specific systems, many adaptations for cold climate PTSs have been examined and case studies illustrating the implementation of cold climate PTSs are detailed in Section 3.2.

3.1. Cold climate effects

Seasonal variations occur in most PTSs due to changes in water flow, senescence of aquatic plants, metabolic activities of microorganisms and variations in temperature and pH. Some wetlands may lose efficiency during the winter months while still meeting water treatment objectives (Groudev et al, 2008). In other cases, no seasonal variation in effectiveness has been found due to complex relationships between oxidation, temperature and pH which maintain the treatment regime regardless of climatic conditions (Hedin, 2013). Some of the main cold climate conditions potentially affecting PTS effectiveness are discussed below.

3.1.1. Temperature

Water temperature changes diurnally and annually. Diurnal temperature changes by approximately 5°C, while annual changes vary depending on latitude and climate. For example, in warm seasons water temperature is approximately equal to air temperature. During periods of ice cover, water temperature is maintained near 0°C for the duration with no diurnal variation (Kadlec & Reddy, 2001). Water temperature directly affects microbial activity by reducing metabolic reaction rates (Oberholster et al, 2014). In addition, plant matter decomposition and adsorption has been shown to decrease as temperature is lowered (Kadlec & Reddy, 2001). The solubility of oxygen (Kadlec & Reddy, 2001) and carbon doxide (Hedin, 2013) is also strongly temperature dependent with greater solubility at lower temperatures. The increased oxygenation in winter can aid aerobic processes, although ice cover can impede oxygenation during the winter and create anoxic conditions (Fortin et al, 2000). Maintaining oxic conditions in sediments during winter has been recommended to increase Fe and Mn retention (Goulet & Pick, 2001). In a PTS in Pennsylvania, Hedin (2013) found there was greater mixing of unfrozen ponds in winter due to density gradients in the water column. Cool water on top continually sunk to the bottom which forced the CO_2-rich deep water to the surface where it degassed. This caused changes in pH and subsequently Fe oxidation rates (Hedin, 2013). This PTS is discussed in greater detail in Section 3.2.10. In general, lower temperatures in winter have the effect of slowing down both biotic and abiotic processes that are essential to the functioning of PTSs, however, in many cases temperature and efficiency are not directly related. For example, at Silver King mine, Yukon, Canada samples were collected with distance from an adit into a natural wetland/muskeg in the months of December, January and February, when temperatures ranged from -26°C to -49°C (Sobolewski, 2003a). In all three months Zn concentrations dropped from 1.2 mg/L at the adit to 0.15-0.61 mg/L within 160 m from the adit into the wetland. Diagnostic sequential leach analysis was used to identify metal removal processes during these winter months and both Cd and Zn were found mostly adsorbed onto or coprecipitated with iron and manganese oxides (Sobolewski, 2003b). More discussion on cold climate impact on microbial activity is presented in 3.1.3.

3.1.2. pH

pH is very important in PTSs treating mine-impacted water as it affects the solubility, oxidation, adsorption and biological removal of metals (Sheoran & Sheoran, 2006). The solubility of many metals is pH-dependent, with metal solubility typically increasing to toxic levels at lower pH values (Figure 5). For certain metals, such as aluminum or iron, raising water pH to neutrality is usually sufficient to decrease their concentrations to acceptable levels. Other metals, such as cadmium, manganese or zinc, can be soluble at neutral pH and cannot be removed by simple pH adjustment. The solubility of certain contaminants, such as nitrate or selenium, is largely unaffected by pH. Finally, the solubility of certain gases, particularly carbon dioxide, is pH-dependent and changes in solution pH will affect their concentrations. CO_2 degassing has been found to increase at colder temperatures due to greater mixing of the water column (Hedin, 2013) and as CO_2 is released, pH increases. CO_2 degassing has also been found to improve iron and trace metal removal in PTSs likely due to the increase in pH (Nairn, 2013). Conversely, CO_2 solubility also increases at lower temperature which may have the effect of lowering pH.

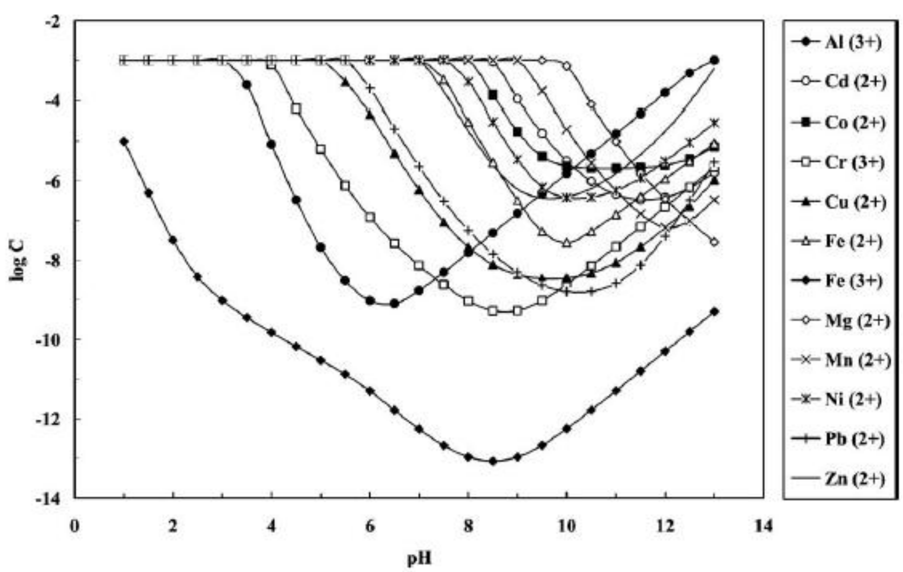

Figure 5: Solubility of different metal hydroxides based on chemical equilibrium calculations using the MINQL+ software. Stimulation conditions: T=25°C, $CI_i = 10^{-2}$ M and $(metal)_i = 10^{-3}$M (Blais et al, 2008, p.137).

Additionally, the precipitation of metals salts, such as aluminum or iron hydroxides can drive significant changes in pH. In some cases, the oxidation and precipitation of dissolved iron may be sufficient to acidify mine water to unacceptable levels. The alkalinity and acidity of water are essential parameters to consider in predicting overall changes in pH that might occur in PTSs.

Biological processes, such as the decomposition of organic matter and plant uptake of metals, are affected by water pH. The sulphate-reducing bacteria (SRB) that are responsible for metal removal, grow within a pH range of 5.5-8.5. Therefore, low water pH hinders their growth and ability to remove metals. Lowering pH has been shown to decrease sulfate reduction rates (Tsukamoto et al, 2004), which is why many PTSs employ pretreatment by ALDs or other limestone-based unit processes. Photosynthesis and respiration have the effect of increasing pH due to the removal of CO_2, therefore pH is likely to be greater in the summer due to growth of algae and plant biomass. pH has been found to have a positive effect on the bioavailability of some metals to some plant species (Deng et al, 2004). Numerous studies have found that temperature and seasonal variation affect pH in PTSs, however the relationship between temperature and pH is complex and requires further study.

3.1.3. Microbial activity

In general, microbial activity decreases with temperature. However, cold-adapted SRB have been studied extensively in the Arctic and Antarctic (Camenzuli et al, 2014; Robador et al, 2009; Sagemann, et al, 1998) and many have adaptations that sustain sulfate reduction rates at colder temperatures. Anaerobic bacteria have been found to grow very slowly below 15°C (Hiley, 2003), although in a CW in Ontario, SRB were found to thrive during winter. Goulet and Pick (2001) showed that the largest SRB populations in a wetland were in December when the water was ~1°C and determined that sulfate reduction rates were highest under those conditions. While sulfate reduction rates are generally lower at colder temperatures and cold-adapted SRB may have higher optimum growth temperatures, sulfate reduction rates may still be maintained at levels similar to those in temperate climates (Fortin et al, 2013). Cold-adapted SRB in the Arctic have been found to be more limited by organic substrate availability than by temperature (Sagemann et al, 1998). Fortin et al. (2013) suggest that decreased organic decomposition in cold temperatures caused the SRB to compensate by increasing their numbers, which maintained the sulfate reduction rates throughout the year. Panos et al. (2013) found that temperature has a large influence on the development of SRB; SRB did not develop at temperatures below 6°C. Despite this, Fe removal efficiency was still relatively high indicating other processes, such as adsorption, may be responsible for maintaining metal removal at lower temperatures (Panos et al, 2013). On the other hand, in a bioreactor at the Calliope mine (Section 3.2.8), results suggested that freezing SRB did not affect their activity during the rest of the year (Nordwick et al, 2006). In a pilot bioreactor in Wyoming, sulfate reduction rates were reduced by 20% in the winter when water temperatures were below 1°C, although this was not enough of a reduction to affect the treatment performance of the bioreactor (Gusek, 2002). In a column experiment on the performance of anaerobic bioreactors at low temperatures, Gould et al. (2012) found that Zn removal fell from 99% to 60% when the temperature was lowered from 20°C to 4°C. They compensated for the loss of efficiency by feeding the columns with liquid carbon (glucose and lactate/acetate) and achieved >95% Zn removal efficiency at 4°C (Gould et al, 2012). Liquid carbon sources can maintain microbial activity in bioreactors over the winter months when cold temperatures reduce organic matter decomposition (Sobolewski, 2010).

3.1.4. Water flow

In cold climates, water flow changes seasonally with large increases in flow in the spring when the snow melts and low flows throughout the winter. Drastic changes in flow present challenges to PTS designers. In a natural wetland in the Rocky Mountains, spring snowmelt caused an increase in the flow to the wetland, which decreased the concentration of metals. The wetland was found to be a sink for Zn and Mn in the summer and a source in the winter due to changes in flow conditions and microbial activity, with the greatest disturbances to treatment efficiency occurring during spring snowmelt (August et al, 2002). Higher flows during spring runoff and throughout the summer likely created more exposed mineral surface area and allowed more of the products of weathering to be flushed out. The higher flows, however, were offset by the increased retention of metals in the wetland. Similar flushing effects were experienced at a subsurface bioreactor constructed in a mineshaft in Montana (Nordwick et al, 2006). Oxygenated spring run-off infiltrated through the ground and mobilized historic metal precipitates creating higher metal concentrations in the effluent in the spring. At a bioreactor in Norway, Cu removal efficiency was found to be highest when water flows were low and constant during the summer (Ettner, 2007). Unlike the majority of sites that experience lower flows in the winter, at this site flows were found to increase in the winter, which resulted in a dramatic drop in removal efficiency. These factors must be accounted for in the design of PTSs.

It is clear that seasonal variations affect the efficiency of PTSs but more work is needed to elucidate the specific mechanisms occurring in cold climate PTSs. Understanding of a site's hydrology, climate, precipitation regime and seasonal water flows allows PTS designers to address site-specific limitations in cold climates. In addition, better understanding of seasonal variations and their effects on metal leaching and metal removal mechanisms will improve the efficiency and reliability of PTSs operating year-round in cold climates.

3.2. Cold climate case studies

Numerous PTSs have been constructed in cold climates at various scales. Below are highlighted case studies of PTSs that demonstrate adaptations in their design for year-round operation in cold climates. Some are successful full scale designs, while others reveal some of the obstacles that may be encountered by PTS designers in cold climates.

3.2.1. United Keno Hill Mine, Yukon – Constructed wetland

In 1995, a pilot wetland was constructed at UKHM and monitored over one year to assess the possibility of treating adit discharge with high levels of Zn (30 mg/L) and other heavy metals (Sobolewski, 1996; Burns, 2000). An area below the Galkeno 900 adit was excavated, fertilized and planted with *Carex aquatilis* collected from a nearby uncontaminated wetland. The plants were allowed to establish for approximately one month, then effluent was piped into a lined pond and then into the wetland. Plywood baffles were subsequently added to distribute the flow evenly throughout the wetland (Figure 6). Zinc removal rates reached up to 90% efficiency in the first season (Table 2), however, the concentration never dropped below 3 ppm, which is well above the 0.5 ppm discharge limit. Other heavy metals were below acceptable limits and more than 80-90% of Co, Ni and Mn were removed. Results from this CW indicated that longer hydraulic retention times within the pilot wetland would have increased the Zn removal efficiency. In addition, Sobolewski (1996) suggested that allowing vegetation to more fully establish (>35% was unvegetated) would likely improve efficiency. Metal removal in the CW was directly correlated with sulfate reduction, suggesting that Zn was removed as a Zn sulfide precipitate. Wetland plants were analyzed from the natural wetland and the results suggested that contaminants uptake from the sediments into the plants was limited.

Figure 6: CW at Galkeno 900 adit with plywood baffles to help distribute flow evenly (Burns, 2000, p.38).

The CW at the Galkeno 900 adit was only operational for a short time and never reached a steady state before flow was redirected to a limestone treatment system inside the mine adit. The wetland was, however, subject to further analysis in 1999 (Burns, 2000). The author reported that four years after transplant, *Carex aquatilis* had fully covered the constructed wetlands. Below-ground seepage from a waste rock pile was observed to be travelling to the wetland so samples were taken to determine the levels and distribution of contaminants in the wetland. The metal concentrations in sediments decreased with distance from the seep while the concentrations in plants were relatively consistent, except for the concentration of Zn which decreased with distance from the seep. The data indicates that the wetland continues to attenuate metals in the discharge waters.

3.2.2. United Keno Hill Mine, Yukon – Anaerobic bioreactor

In 2008 at UKHM, a pilot horizontal flow bioreactor was constructed on-site at the Galkeno 900 adit (Figure 7 and 8) (Alexco, 2012). A pit was excavated, lined and then filled with waste rock. The pit contained 3 m deep waste rock with a 1.2 m soil cover on top to reduce ice formation in the bioreactor. In March 2011, there was up to 60 cm of ice in the bioreactor. All of the pipes and valves were buried at least 1 m belowground and those running under roadbeds were covered in heat trace. Baffles were installed to distribute the flow throughout the pit. The bioreactor was filled with untreated adit water and sucrose, methanol and/or milk solids were added to stimulate SRB growth in October 2008 and

January and July 2009. From October to July of 2009, the water was recirculated in the bioreactor in a closed circuit and strong sulfate-reducing conditions developed. The flow-through phase then began and constant additions of methanol were initiated. The flow rate was initially 0.5 L/sec (21 days residence time), which was then increased to 1 L/sec (10.5 days residence time) and then reduced to 0.75 L/sec for the rest of the operational phase. At low flows 99.8% of Zn was removed and at higher flow rates 97.8% was removed (Table 2). While the Zn discharge limit (0.5 mg/L) was met consistently at the low flow rate, it was sometimes exceeded at higher rates. It was observed that the residence time within the bioreactor was a major factor affecting metal removal efficiency. While Zn was the primary contaminant of concern, 80% of As was removed at a flow rate of 1 L/sec. Cd was removed to below detection limits and 97.5% of Ni was removed at low flows, but only ~50% was removed at higher flows. Iron and Mn were efficiently removed (99%) during the recirculation phase, but during the operational phase, when the reducing conditions were fully established, Fe and Mn were remobilized resulting in slow release of these metals over the course of operation. The bed of the bioreactor was composed only of waste rock, so sorption was not likely a factor in removing metals; sulfate reduction and metal sulphide precipitation were the primary mechanisms. Some of the engineering problems encountered included uneven distribution of flow and decreased contact with the substrate (i.e. so-called 'dead zones') and pump failure and power outages resulting in pipes freezing and the system having to be shut-down temporarily. A gravity fed system and/or back-up power would have helped to avoid the difficulties of maintaining a constant flow in the winter (Alexco, 2012).

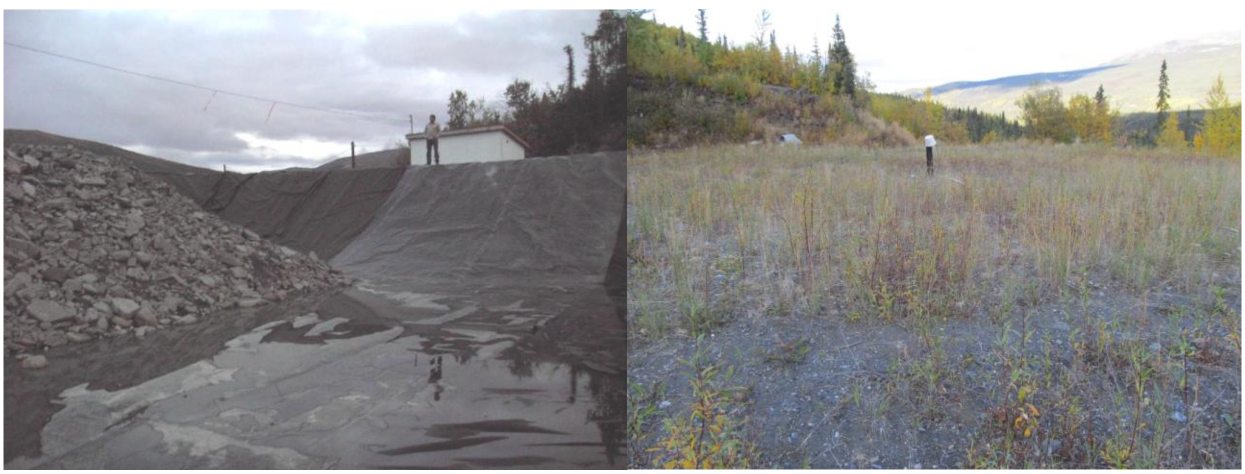

Figure 7: Bellekeno pilot bioreactor under construction in 2008 (left) and after 5 years (right) (Alexco, 2012, p. 6).

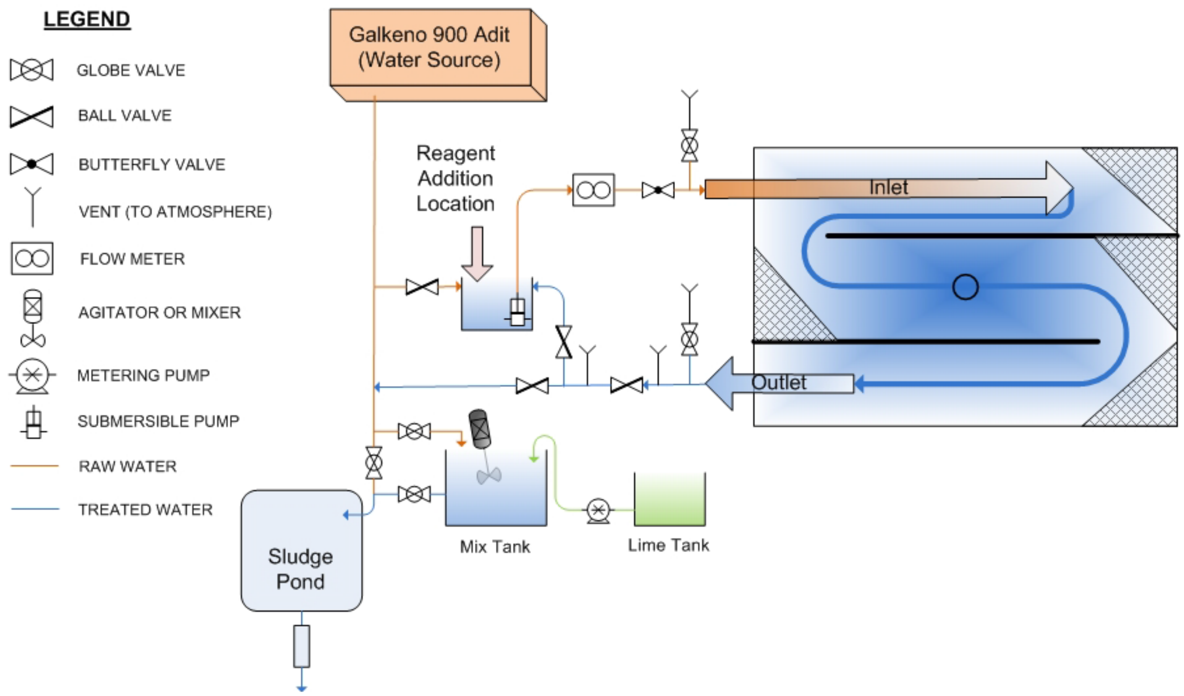

Figure 8: Layout of Galkeno 900 bioreactor (Alexco, 2012, p.7).

3.2.3. Tulsequah Chief & Smoky River Coal Mine, BC

At the Tulsequah Chief Mine (Cu, Pb/Zn) in Northern British Columbia, acidic rock drainage was successfully treated using a 66 m^3 anaerobic bioreactor built inside the mine adit (Sobolewski 2010). Wood mulch was used in the bioreactor; however, as the ambient temperature in the reactor was low (6-8°C), wood mulch decomposition was too slow to maintain anaerobic conditions. To overcome the issue, ethylene glycol was subsequently added to the influent to create reducing conditions, decreasing the dissolved oxygen levels and kick-starting SRB activity. A full scale five cell, 200 m long, bioreactor was then built based on the pilot test results and monitored over four years. The first three cells received a wood mulch/limestone mix to reduce acidity and the two final cells received sulfur prills (elemental sulfur in pellet form) and supported SRB. Ethylene glycol was dripped into cells one and four. The front three cells consistently removed Al and Fe, but also gradually accumulated sludge, which caused failure after four years of operation, while the final cells removed Cd, Cu and some Zn (Table 3). The bioreactor operated year-round without maintenance in the winter and was constructed in areas with limited space, access and power.

Table 3: Metal concentrations in drainage and discharge from Tulsequah Chief pilot and full scale bioreactors (adapted from Sobolewski, 2010).

Parameter	ARD	Pilot-scale bioreactor	Full-scale bioreactor
pH	3.0-3.7	6.0-6.7	6.1-6.7
Al (mg/L)	10-30	0.1-1.0	0.06-2.5
Cd (mg/L)	0.22-0.47	<0.050	0.07-0.30
Cu (mg/L)	16-30	0.1-1.0	2-7
Fe (mg/L)	20-120	0.1-2.0	0.04-4.0
Zn (mg/L)	56-120	30-50	42-77

3.2.4. Smoky River Coal Mine, BC

Two 5 m^3 pilot scale bioreactors were constructed in parallel at the Smoky River Coal Mine to treat Se-contaminated water in an area where access, space and power were severely restricted, (Sobolewski, 2010). The bioreactors were filled with gravel, wood chips, manure, lime and bonemeal, inoculated with Se-reducing bacteria and fed with ethylene glycol as a liquid carbon source. The liquid carbon source helped to sustain necessary microbial processes (Sobolewski, 2010). The bioreactors effectively removed 97% of the Se from September to November 2008 (Figure 9), even when temperatures dropped to 2°C. This case study, along with other studies previously mentioned (i.e. Galkeno 900 and Tulsequah Chief) demonstrates that addition of a liquid carbon source may allow anaerobic bioreactors to keep operating with high efficiency on northern mine sites under winter conditions.

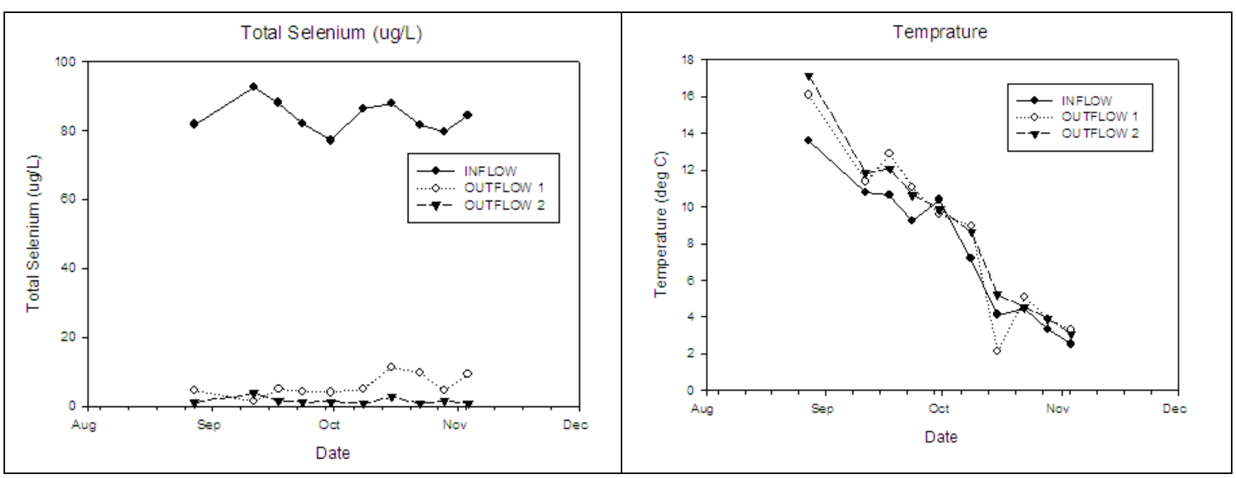

Figure 9: Selenium removal as a function of temperature at Smoky River Coal (Sobolewski, 2010, p. 2)

3.2.5. Kongens & Folldal Mines, Norway

In Norway, bioreactors have been tested for the treatment of acidic mine drainage at both active and abandoned mines in cold climates (Ettner 2007). A first pilot bioreactor system was constructed at the Kongens Mine in northern Norway in 1999, consisting of a pre-treatment limestone drain and four anaerobic cells filled with organic waste materials, rotted hay, sawdust, tree clippings and $CaCO_3$. However, the substrate was found to be inappropriate due to low hydraulic conductivity which caused clogging. Highlighting the importance of thorough design, the substrates were revisited and a new bioreactor system was constructed in 2002 using rotted hay, sand and coarse birch tree materials. In this reactor, up to 85% Cu 48% Zn, 98% Cd, 82% Pb, 71% Cr and 24% Ni were removed in this second pilot system, however Cu removal was significantly reduced in winter months. This reduction was correlated with a drastic increase in the flow rate in the bioreactor (Figure 10). To overcome this issue and avoid high flow variations, a full-scale bioreactor was constructed in 2006 which consists of two identical cells, one receiving a constant flow and the other receiving smaller, but variable flow rates.

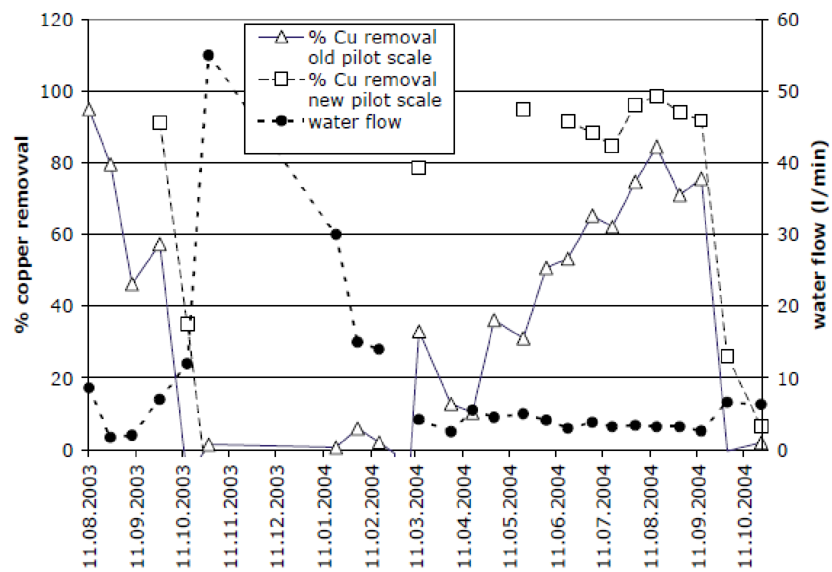

Figure 10: Copper removal (%) by pilot anaerobic reactors versus water flows (Ettner 2007, p.3)

Using a similar design, a pilot bioreactor system was built at the Folldal mine in 2006, with four 8 by 1.5 m basins filled with rotten hay, tree clippings, shell sand and gypsum to act as an extra sulfate source (Ettner 2007). The bioreactors were filled with water in the fall and left to rest for the winter with no flow. Unfortunately, no monitoring data were available from when system was restarted in the spring. However, the study indicated that cold winter temperatures did not prevent inoculation of the bioreactors by microorganisms and development of anaerobic conditions, low Eh and signs of sulfide precipitates were reported.

3.2.6. Bell Copper Mine, BC

At the Bell Copper mine in Smithers, BC, two experimental wetlands were constructed to determine the effectiveness of a year-round CW in a northern environment for treatment of Cu-contaminated water (Sobolewski et al, 1994). Two membrane lined ponds (300 m^2 and 75 m^2) were constructed, fertilized with manure and planted with peat mats dominated by sedges (*Carex aquatilis* and *Carex laeviculmis*). The ponds were also planted with cattails (*Typha latifolia*) and beaked sedges (*Carex rostrata*) and were fully vegetated within a year (Figure 11). For the first year, the wetlands were fed with low flows of mine drainage with low levels of Cu (0.3-1 ppm) at a pH of 6-8. They were 98% effective at removing Cu (Figure 12) throughout the winter months when the wetlands were covered in ice and even when the influent Cu concentrations occasionally spiked to 10 ppm (Sobolewski et. al., 1994). Mine drainage with high levels of Cu (35-50 ppm) at pH 3.5 was then introduced to the wetlands. For the first six weeks, they maintained removal efficiency, but then gradually started deteriorating reaching 40% efficiency for the large wetland and 80% for the small one. Both a decrease in pH and SRB population were observed concurrently with the reduction of Cu removal efficiency. The reduction of efficiency of the wetland was likely due to reduction in the activity and growth of SRB, which were not acclimated to the low water pH. Longer retention times and pretreatment to increase pH were suggested as means to address the shortcomings of these wetlands for treatment of highly contaminated mine drainage (Sobolewski et al, 1994).

Figure 11: Pilot wetlands at the Bell Copper Mine (Sobolewski A., personal communications, November 12, 2014).

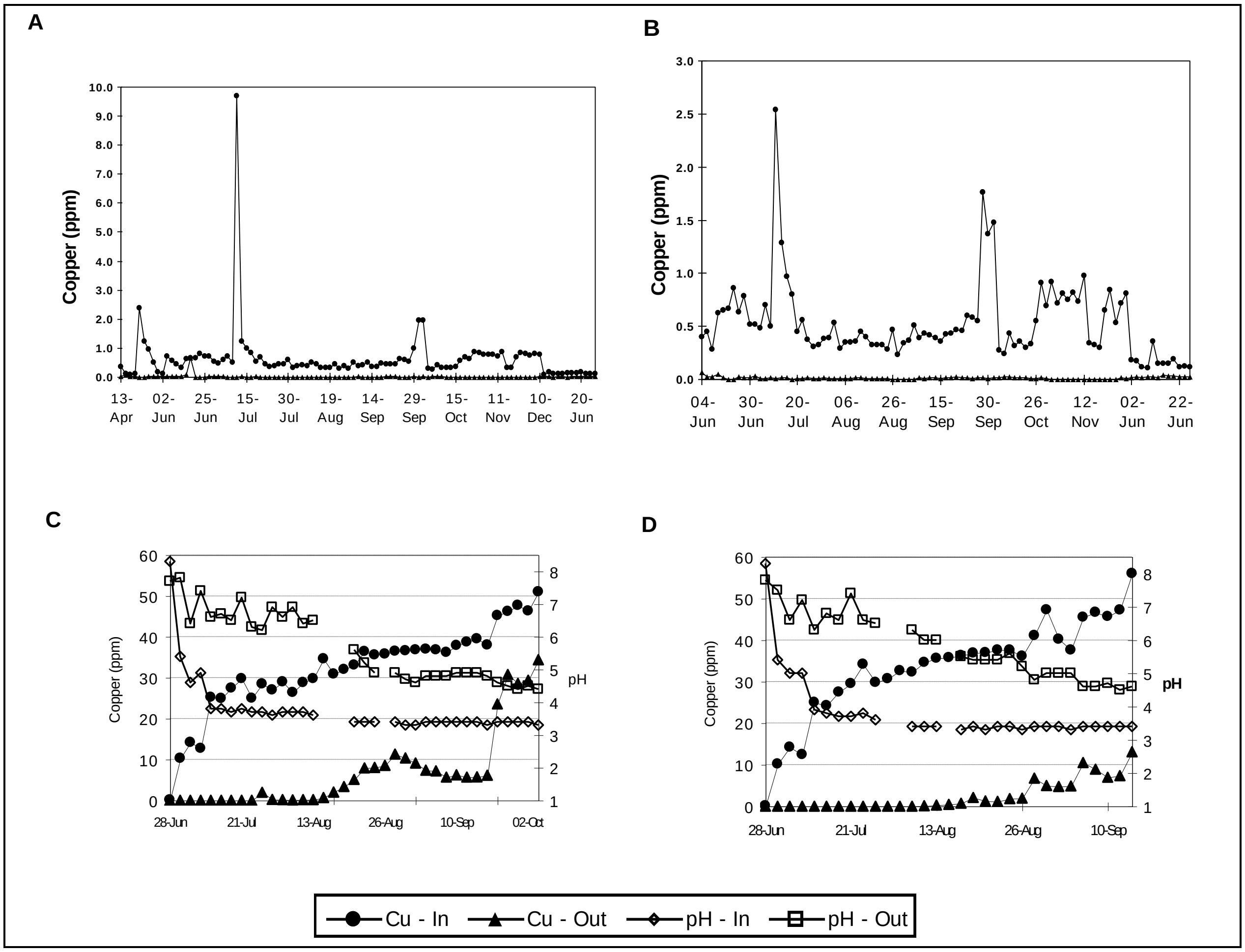

Figure 12: Copper concentrations and pH of influent and effluent for the large (A, C) and small pond (B, D) at low strength (A, B) and high strength feed (C, D) (April 1992 to Oct 1993, Sobolewski et al, 1994, p. 686). Note different scales used in A and B.

3.2.7. Lilly/Orphan Boy Mine, Montana

At the Lilly/Orphan Boy Mine in Montana, USA an in-situ bioreactor was constructed and operated between 1994 and 2005 to treat ARD at the abandoned mine (Canty, 2000; Nordwick et al, 2006; Bless et al, 2008). The bioreactor was built within an old mineshaft, approximately 9 m below the static water level. Organic matter, cow manure and straw were placed on a platform inside the shaft with mine water flowing upward through the organic substrate (Figure 13). This below-ground design offered protection from the cold winter temperatures (ambient temperature was 8 °C) and worked successfully for over 11 years. Removal efficiencies were between 98% and >99% for Al, Cd, Cu and Zn and 96% for As. However, the mineshaft bioreactor design also came with challenges, including metal leaching within the mine tunnel as water travelled out through the portal, creating effluent with, in some cases, higher contaminant concentrations (As and Fe) (Table 4). Additionally, this system was affected by snowmelt and spring run-off. Higher metal concentrations in the spring were attributed to greater metal leaching from the rocks by oxidation due to oxygenated surface water infiltrating into fractures in the rocks and tunnel walls (Nordwick et al, 2006). For most metals, metal removal had seasonal variability with high removal efficiencies during the summer, but low removal efficiencies at all other times of the year (Figure 14). However, Mn and Zn were removed effectively year-round.

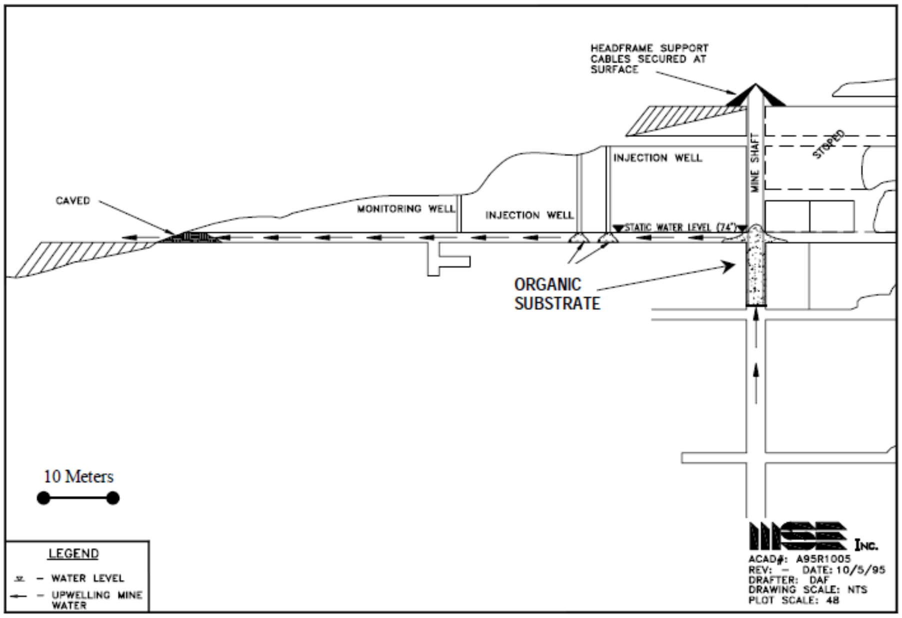

Figure 13: Underground bioreactor at Lilly/Orphan Boy Mine, Montana (Nordwick et al, 2006, p. 1412).

Table 4: Representative mine water chemistry at Lilly/Orphan passive treatment system (dissolved metals in mg/L) (Bless et al, 2008, p. 243).

Sample	Month	Fe	Zn	Al	Mn	As	Cd	Cu	SO_4	pH
Pretreatment shaft	Several	27.7	26.1	9.7	6.2	1.07	0.33	0.32	277	3.0
Pretreatment portal	Several	14.1	19.4	7.4	5.5	0.08	0.24	0.33	213	3.4
Treated tunnel water	March 2001	9.3	<0.01	<0.02	1.4	0.02	<0.005	0.022	59	7.4
	May 2001	13.2	0.06	<0.02	2.7	0.13	<0.005	<0.005	214	7.0
Treated portal effluent	March 2001	30.8	10.1	0.2	5.5	5.26	0.040	<0.002	161	6.1
	May 2001	11.2	28.8	11.2	6.4	0.05	0.301	0.686	394	3.4

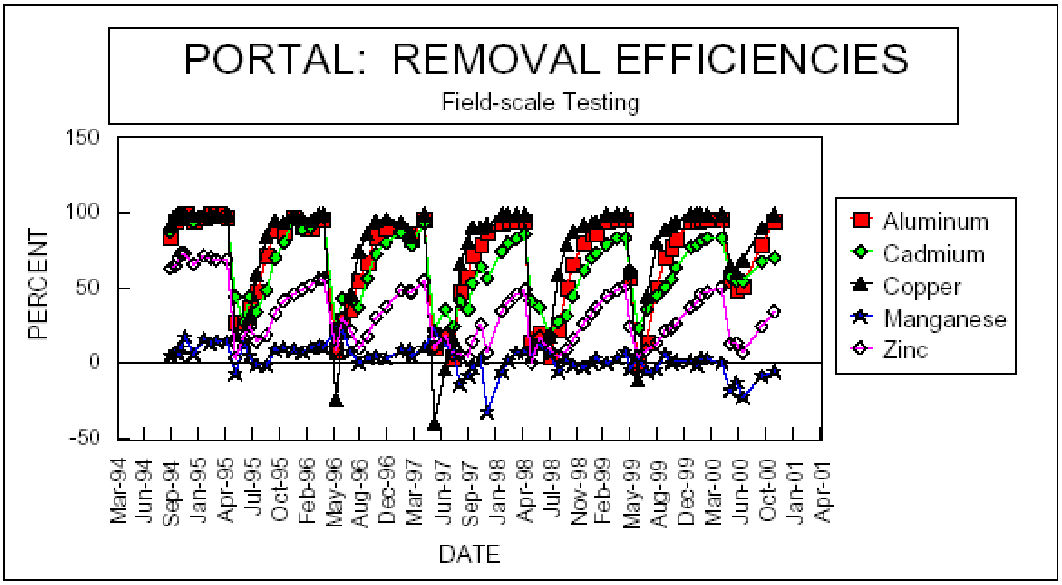

Figure 14: Metal removal in Lilly/Orphan Boy Mine (Canty, 2000, p. 3).

3.2.8. Calliope Mine, Montana

Pilot-scale bioreactors were built at the Calliope Mine in Montana to determine the effect of freezing on SRB (Nordwick et al, 2006; Wilmoth 2002). Two bioreactors were placed in trenches (21.8m and 18.6m long) and one was constructed aboveground (22.1m long) to investigate the impact of seasonal freezing and thawing on treatment performance (Figure 15). The trenches were filled with cow manure, straw, limestone and Terracell^{TM} to prevent organic matter settling, improve permeability and ensure uniform flow through the bioreactors (Figure 16). The systems were operated with a horizontal flow and 4.5 days residence time (+1 day pretreatment in the longer belowground system). Once the SRB populations were established, the mine water was introduced. Zinc, Cu, and Cd were removed as sulfides through bacterial activity while Fe, Mn, Al and As removal was affected by SRB only in an indirect manner by responding to increased pH caused by SRB activity. Zn, Cu and Cd were removed from the ARD to approximately 500 to 800 µg/L for Zn, 80 µg/L for Cu, and 5 µg/L for Cd in both the belowground and aboveground configurations, indicating that at Calliope site, the bioreactor efficiencies were not affected by freezing and thawing. Removal of Fe, Al and Cu was very effective (99%), whereas As was not removed at all and Mn were removed to a lesser extent (32%). Plugging by chemical precipitates affected the flow rate during operation. However it was observed after deconstruction of the aboveground bioreactor that less organic material would have been just as effective, which would have increased the permeability without compromising the efficiency. Overall, a uniform horizontal flow was successfully achieved in the bioreactors. The pilot project also demonstrated that SRB need time to establish, but once established and supplied with organic carbon, they can maintain an active population at temperatures ranging from 2°C to 16°C, without being affected by winter conditions (Figure 16). Additionally, a well-established population can withstand freezing in winter with little or no effect on their activity during the rest of the year.

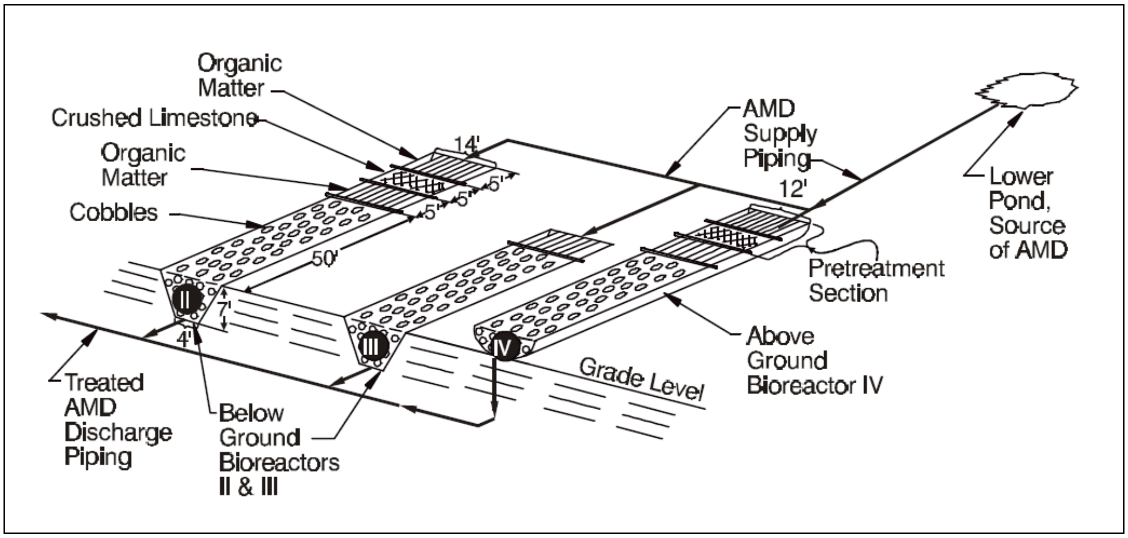

Figure 15: SRB bioreactors at Calliope Mine (Wilmoth, 2002, p.21).

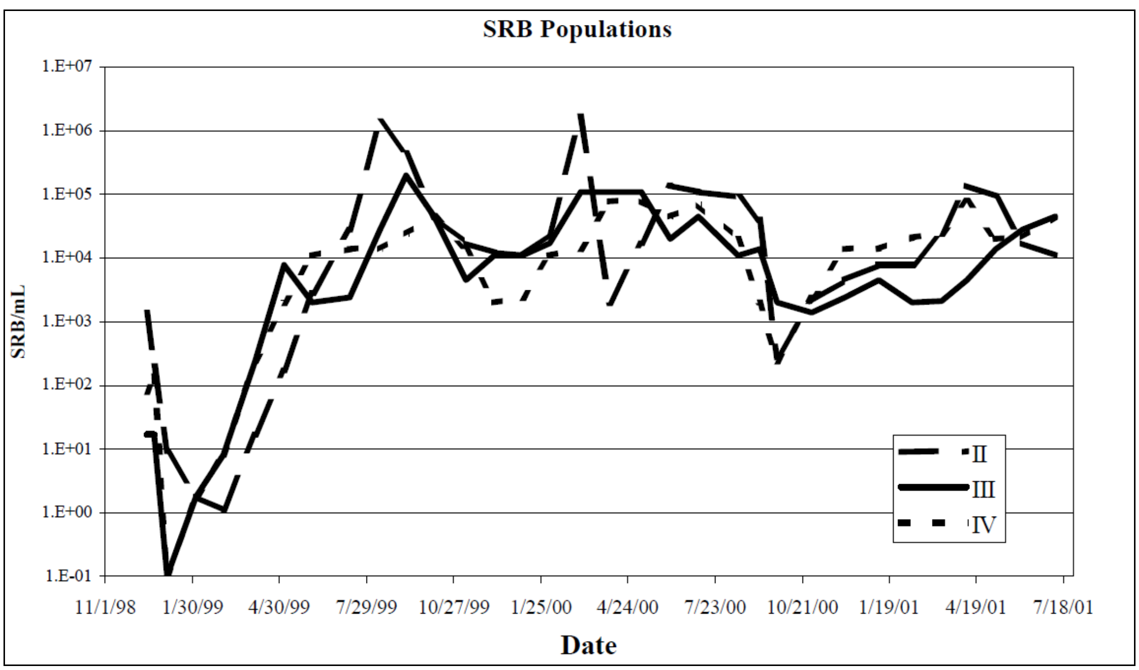

Figure 16: SRB population variation over 3 years in below-ground (II and III) and above-ground (IV) bioreactors at Calliope mine, Montana (Wilmoth, 2002, p. 35).

3.2.9. Cadillac Molybdenite Mine, Quebec

At the decommissioned Cadillac Molybdenite Mine in Northern Quebec a 170 m^3 bioreactor was constructed to treat ARD with acidic pH ranging from 2.7-5.6 (Kuyucak et al, 2006). Passive treatment was chosen due to the lower cost and maintenance as compared to an active system and the remoteness and limited access for much of the year. The extreme cold for 6 months of the year (average -20°C) also presented a challenge in the design of the system. The treatment unit consisted of a collection ditch, an anaerobic cell filled with wood chips, limestone, hay and manure, followed by an oxidation pond and a limestone drain (Figure 17). The nutrient mixture was allowed to condition for 2 months prior to installation in the cell to allow the establishment of SRB before winter. In addition, an overflow drain was installed in the anaerobic cell and the collection ditch to allow excess mine drainage to bypass the SRB cell and help keep uniform flow in the bioreactor. The surface of the collection ditch and anaerobic cell were covered with 50 mm polystyrene sheets and 0.25 m of clay to prevent freezing (Figure 18). During the winter, the effluent temperature dropped to 3°C and the anaerobic cell was at 4°C. Since its implementation in 2004, removal to below regulated limits for Cu (0.3 mg/L), Ni (0.5 mg/L) and Zn (0.5 mg/L) has been successful (Table 5). Iron removal was not as successful during the winter, when settling of Fe precipitates in the oxidation pond was inhibited by freezing. Aluminum removal was also improved during the summer as SRB activity increased. Monitoring of the site continues and an expansion is planned to accommodate all of the effluent.

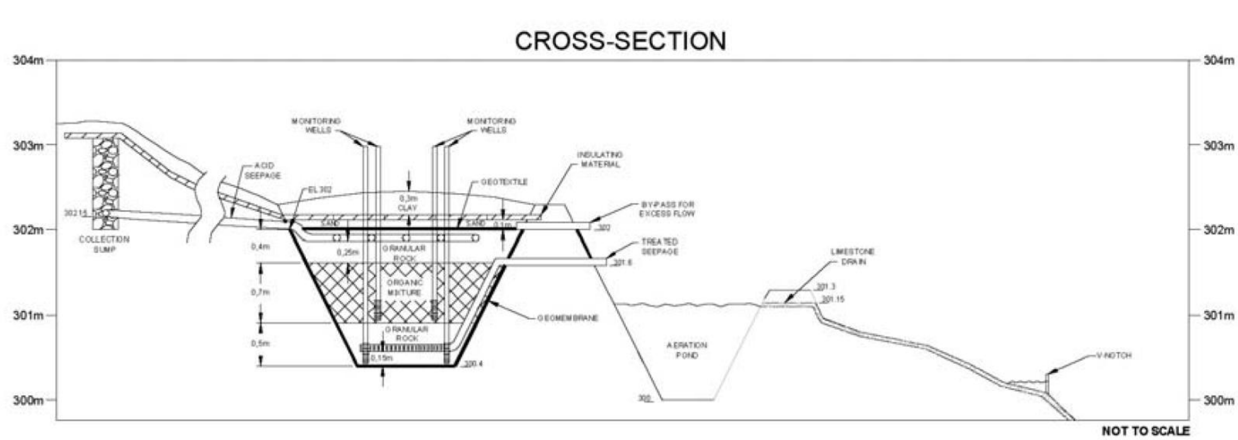

Figure 17: Bioreactor design at the Cadillac mine in Quebec (Kuyucak et al, 2006, p. 988).

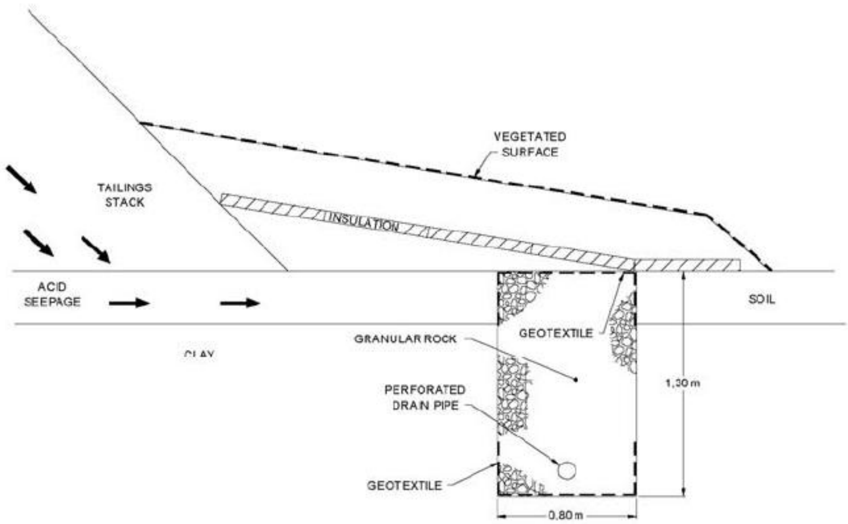

Figure 18: Cross-section of the collection ditch at the Cadillac mine PTS illustrating the use of insulation above the seepage (Kuyucak, et al, 2006, p.989).

Table 5: Water quality before and after passive treatment system at Cadillac mine, Quebec (from Kuyucak et al, 2006, p.982)

Parameter *	Directive 019	Before Treatment Average *	After Treatment Nov'04	Dec'04	Jan'05	Feb'05	Mar'05	Apr'05	Average 2005
pH	6.5	3.45	6.4	6.3	6.3	6.3	6.4	6.6	6.7
TSS	25	230	32	3.5	17	22	14	17	11.7
Al	-	43	-	-	-	18	19	11	9.4
Cu	0.3	0.3	0.1	0.06	<0.01	<0.01	<0.01	<0.01	0.008
Fe**	3	13.5 (32)***	10.5	11	2.8	1	0.6	0.3	0.12
Mn	-	5.8	-	-	-	5.7	5.5	3	3.5
Ni	0.5	0.6	0.33	0.18	0.04	0.02	0.02	<0.01	0.01
Zn	0.5	1.35	0.48	0.3	0.02	0.02	0.02	0.02	0.012
SO_4	-	887	798	690	700	620	500	360	-

* All in mg/L except pH
** >90% of iron was in Fe(II) form
*** average in 2005

3.2.10. Marchand Mine, Pennsylvania

In a PTS treating coal mine drainage in Pennsylvania, Hedin (2013) found that seasonality did not significantly affect Fe removal efficiency over a six year monitoring period. The system is comprised of 6 connected ponds and a CW planted with a mix of aquatic species (Figure 19). Models predicted Fe oxidation rates would be 10 times slower in cold seasons than in warm seasons (-15°C to +35°C); however, that was not the case in this PTS; Fe removal was found to be temperature independent (Figure 20). An inverse relationship between pH and temperature offset the expected reduction in efficiency. CO_2 solubility increases as temperature decreases, so it was unexpected when CO_2 degassing increased at colder temperatures in the ponds. This was likely due to more thorough mixing at colder temperatures. Surface waters are continually cooled and sink due to density gradients resulting in deep water that is more saturated with CO_2 coming to the surface and degassing. The increased degassing causes an increase in pH, which in turn increases Fe oxidation rates. Although this process can account for the results observed in the ponds, the seasonal dynamics in the wetland cell were far more complex. While the temperature in the oxidation ponds never fell below 4°C, portions of the wetland froze during cold weather and it was largely covered with ice during cold periods. The ice cover reduced gas exchange processes during the winter and biological activity increased during the summer. As a whole, the PTS was effective year-round at reducing Fe levels to below permissible limits (Table 2) and Fe removal efficiency did not change seasonally. Additionally, the iron sludge precipitated in the ponds is recoverable and may be sold to offset the treatment costs (Hedin, 2013). This case study highlights the complex interactions that can occur between temperature-sensitive processes, complicating predictions of temperature effects on removal rates.

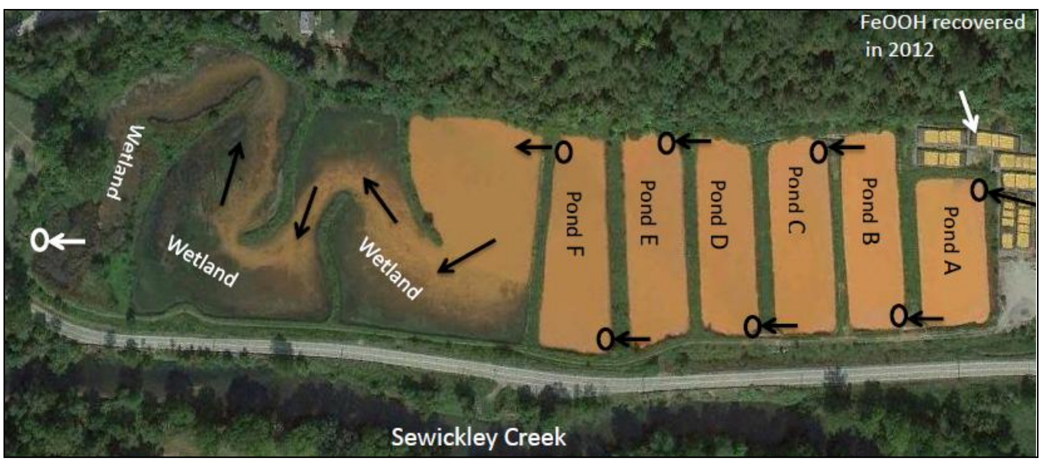

Figure 19: Aerial view of the Marchand mine bioreactor ponds, wetlands and recovered iron sludge (Hedin, 2013, p.9).

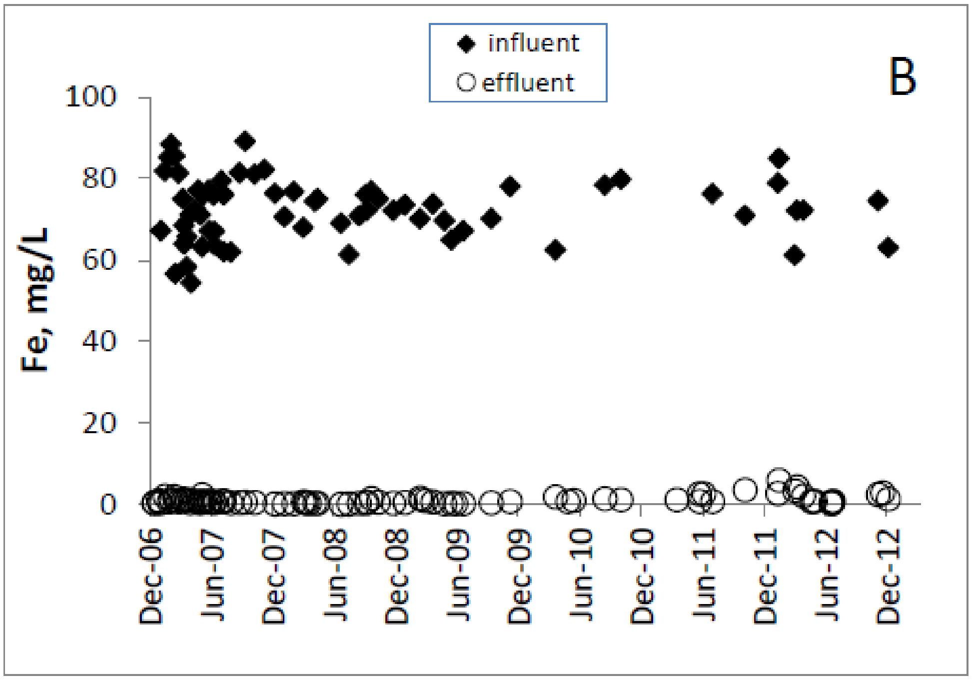

Figure 20: Total Fe concentration at influent and effluent of the Marchand mine treatment system, Pensylvania (Hedin, 2013, p.10).

3.2.11. Teck Smelter, Trail, BC

A PTS was constructed near Trail, BC to treat effluent containing very high concentrations of As, Zn and Cd from a Pb/Zn smelter (Table 6) (Duncan et al, 2002; Mattes et al, 2004). The first phase, constructed in 1997, consisted of three wetland cells (50 m^2 for 1st and 2nd cell and 300 m^2 for 3rd cell). The first cell was planted with *Brassica spp*. and *Helianthus annuus*, the second planted with *Calamagrostis canadensis* and other native grasses from the local area and the third cell was planted with *Typha latifolia* (Figure 21). The cells were initially watered with stream water to allow root growth to establish during the first year. In 1998, a vertical sub-surface flow bioreactor was built upstream with pulp mill biosolids, sand and cow manure. It was designed to allow 1 m of standing water to sit on top to protect it from winter freezing and to ensure it stayed anaerobic. Additionally, all pipes and control valves were buried below 0.9 m. Due to plugging in the first bioreactor and to increase Zn removal, another bioreactor cell was similarly constructed in 2000, upstream of the first (Figure 21), which was subsequently rebuilt. The second cell included limestone to increase the pH and thus improve Zn removal efficiency. The system was fully winterized for year-round operation, although was shut down due to a frozen pipe after a year of operation. After two years of operation (spread over a 5 year period), the system reached a net removal efficiency of 96% for As, Cd and Zn (Table 6), although As levels in the effluent were still not below stringent discharge limits. While seasonal efficiencies are not available, this system purportedly works efficiently year-round and successfully withstands freezing conditions.

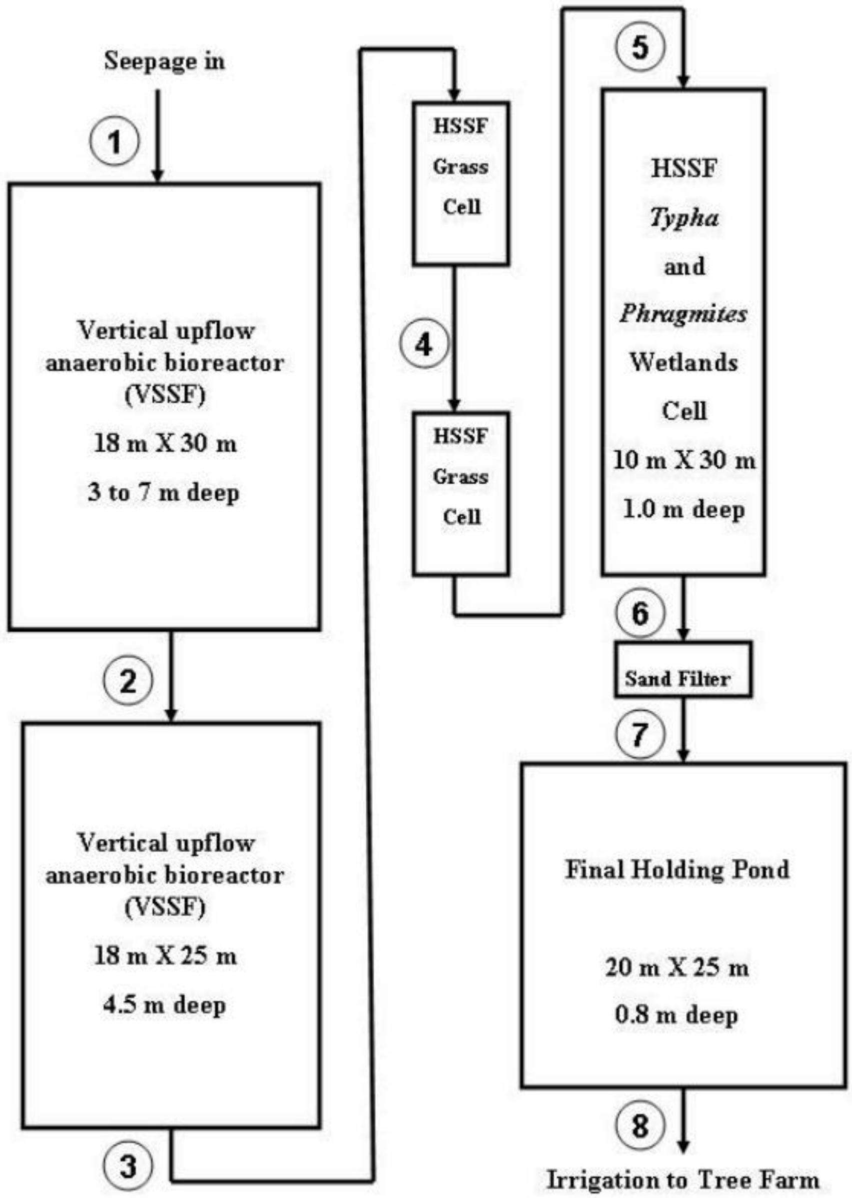

Figure 21: Diagram of the sequence of PT components at the Teck smelter, Trail, BC (Duncan, 2010, p. 40).

Table 6: Metal concentrations (ppm) and percentage removal during to operating periods (257 and 576 days) of the passive treatment system at Teck smelter, Trail, BC (Duncan et al, 2002, p.5).

Sample Point	As. Conc.	% Removed	Cd. Conc.	% Removed	Zn Conc.	% Removed
Input	69.5		5.1		355	
Anaerobic 1	18.7	73.1	2.1	58.8	177	50.0
Anaerobic 2	8.2	56.1	0.4	81.0	101	43.4
Cell 1	5.0	39.0	0.1	75.0	55.4	45.2
Cell 2	4.7	1.5	0.1	0.00	46.8	15.5
Cell 3	1.5	68.1	0.04	60.0	19.8	57.7
Final Pond	1.3	13.3	0.03	25.0	15.1	23.7
Total %		98.1		99.4		95.7

3.2.12. Uranium mine, Curilo, Bulgaria

A four cell constructed wetland in Curilo, Bulgaria successfully treated acidic drainage (pH 2-4) from a uranium deposit over a 10 year period (Groudev et al, 2008). The wetland cells were lined with a mixture of soil, plant and spent mushroom compost, cow manure, crushed limestone, silt and sand. Several barriers of this mixture were placed perpendicularly to the water flow. The wetland was vegetated with *Typha latifolia*, *Typha angustifolia, Phragmites australis, Scirpus lacustris,* and some *Juncus, Eleocharis, Poa* and *Carex* species. The wetlands significantly reduced heavy metal concentrations year-round, operating with a flow-rate between 0.2 to 1.0 L/s and a residence time between 0.5 and 5 days (Table 7). While efficiency was strongly temperature dependent, the effluent metal concentrations during the winter months (when water temperatures were below 5°C) were still below permissible discharge concentrations. Microbial activity was lower in wintertime; however, the longer residence time (i.e. lower flow) helped to sustain satisfactory removal efficiency. High metal concentrations measured in dead biomass indicated that sorption processes by dead plants and clays played an important role in removing metals in during the winter (Groudev et al, 2008).

Table 7: Water quality data before and after treatment by constructed wetlands at Curilo, Bulgaria (Groudev et al, 2008, p.94).

Parameters	Before treatment	After treatment	Permissible levels for waters used in agriculture and industry
Temperature, °C	(+0.1)–(+25.1)	(+0.1)–(+28.0)	–
pH	2.15–4.15	6.51–7.92	6–9
Total dissolved solids, mg/l	590–3250	341–2804	1500
Solids, mg/l	23–170	15–122	100
Dissolved organic carbon, mg/l	0.3–4.6	18–88	20
Sulphates, mg/l	341–1784	280–1254	400
Uranium, mg/l	0.28–4.82	<0.10	0.6
Radium, Bq/l	0.05–0.55	<0.05	0.15
Copper, mg/l	0.41–14.9	<0.1–<0.5	0.5
Zinc, mg/l	0.80–24.2	<0.5	10
Cadmium, mg/l	0.01–0.12	<0.01	0.02
Lead, mg/l	0.10–0.82	<0.10	0.2
Nickel, mg/l	0.32–7.35	<0.1–<0.5	0.5
Cobalt, mg/l	0.23–6.20	<0.1–<0.5	0.5
Iron, mg/l	73–1072	<1–14	5
Manganese, mg/l	0.95–55	<0.5–6.4	0.8
Arsenic, mg/l	0.01–0.59	<0.01–<0.1	0.8

3.2.13. Wood Cadillac, QC

An anaerobic bioreactor was built at the historic Wood Cadillac mine site, in northwestern Quebec, to treat an As-rich effluent. The tailings at Wood Cadillac are sulphidic with an excess of buffering capacity, producing neutral mine drainage (Tasse et al, 2003). The 50 x 57 x 1 m bioreactor was filled with local yellow birch bark to create reducing conditions and fed with seepage at average flow rates of 7.3L/sec in 2000 and 5.0 L/sec. in 2002 (Germain and Cyr, 2003). The biofilter displayed high As removal efficiencies, even in November with a typical seasonal temperature decrease (Figure 22) and the concentrations of As were lowered to below 0.057 mg/L except in June 2001. Overall, this bioreactor fulfilled its objectives over the first three years when it was monitored. Later in 2004, after 5-years of operation and with no interventions, Libeiro (2007) studied the bioreactor again and reported that As removal was still achieved by the reactor, lowering the inlet concentration of 0.03-0.26 mg/L down to 0.002-0.05 mg/L at the bioreactor outlet.

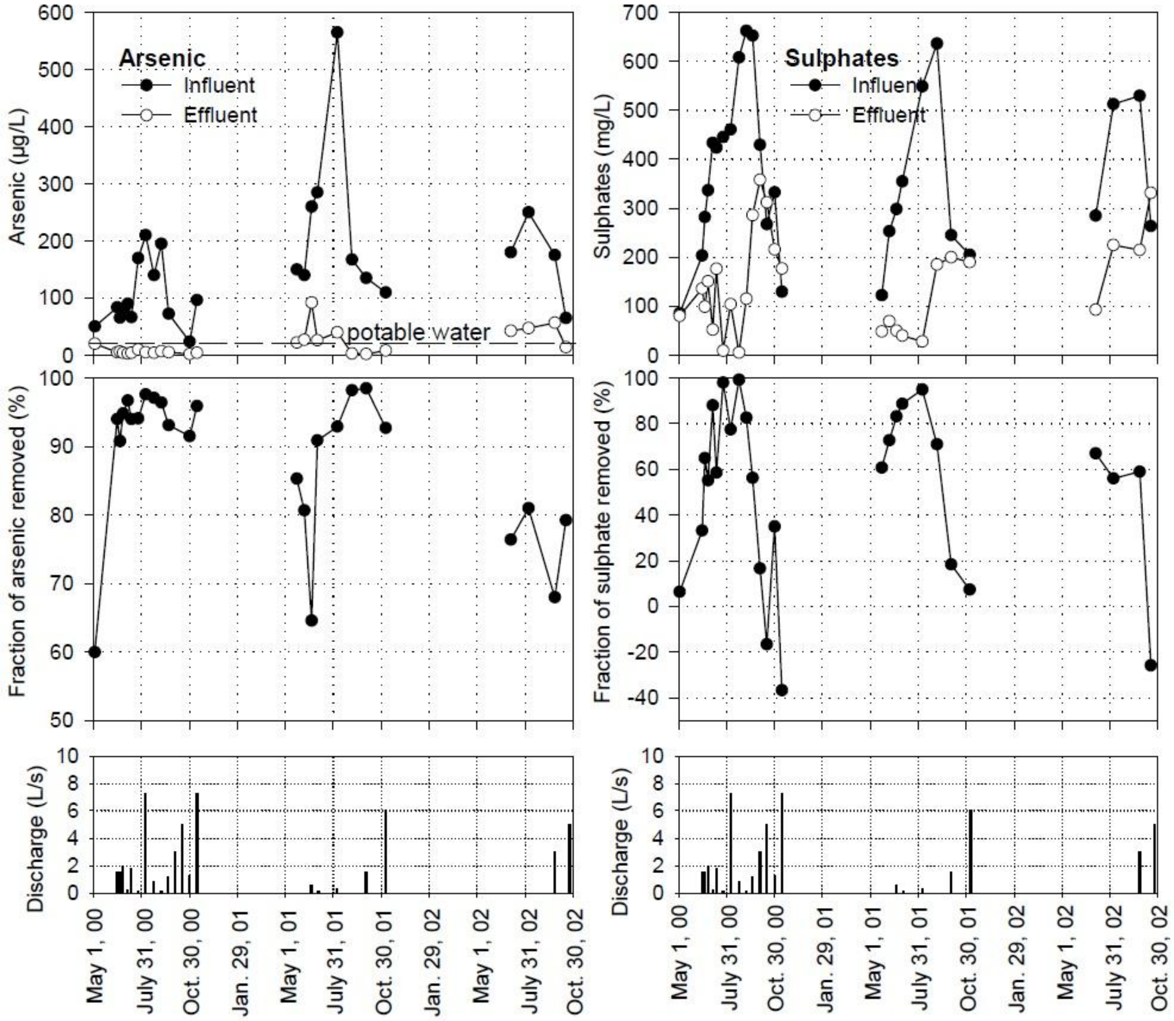

Figure 22: Arsenic and sulphates at the influent and effluent of the biofilter and their respective fraction removed from May 2000 to October 2002. Measured discharge rates are also shown (Germain and Cyr, 2003, p. 7).

3.3. Cold climate solutions

Several pilot scale and full scale PTSs have been built in cold climates and while the efficacy of these PTSs varies, many have successfully treated mine-impacted water (Sobolewski et al, 1994; Nordin, 2000; Duncan, 2002; Wilmoth, 2002; Kuyucak et al, 2006; Groudev et al, 2008; Sobolewski, 2010; Alexco, 2012; Hedin, 2013). It is clear, however, that cold climate adaptations are required for PTSs in these environments. While site-specific adaptations are often required, some basic principles are common to most cold climate PTSs, discussed below.

3.3.1. Burial and insulation

Freezing temperatures cause obvious problems where liquid water is concerned. Solutions are also relatively straightforward. Freezing of PTSs due to cold ambient winter temperatures can be avoided by underground burial or belowground operation. To avoid hydraulic failure caused by freezing, pipes should be buried below the frost line and/or heat-traced (Duncan, 2002; Alexco, 2012). Bioreactors may also be built in abandoned mineshafts to allow year-round operation in cold climates (Nordwick et al, 2006).

Insulating PTSs from freezing can also be achieved in various ways. Insulation may be in the form of burial under organic matter, vegetation or standing water, as in the case of horizontal sub-surface flow wetlands (Duncan et al, 2002); insulation with a layer of straw (Kadlec et al, 2003); polystyrene sheets (Kuyucak et al, 2006); air pockets (Giaever, 2003); or snow cover (Kadlec et al, 2003). For systems in the far north, the design must account for interactions with permafrost, both in the short-term (potential freezing from the sides and bottom of excavations) and in the long-term (subsidence induced by loss of permafrost). If precautions are taken then maintenance of above zero temperatures in bioreactors or CWs can be relatively easy, as decomposition and bacterial activity help to maintain temperatures.

3.3.2. Establishment of biotic communities

PTSs rely on biotic components for which seasons of high productivity are often short and also coincide with periods when construction takes place. Bacteria and vegetation in northern environments are adapted to survive and thrive in these conditions, however, in engineered systems the time needed to establish these communities can be overlooked. Due to extreme seasonal variation in cold climates, any PTS with components of vegetation or microorganisms must prepare for periods of low or no productivity. Construction should be planned such that vegetation is allowed to fully establish in wetlands before mine-impacted water is introduced (Goulet & Pick, 2001). Typically, excavation of beds and other earthworks will be done in the fall, while plant propagation and fertilization is done in early spring, allowing a full growing season to recover from transplantation shock and establishment of a sturdy root system.

Similarly, SRB populations need time to colonize substrates prior to introduction of contaminated waters and cold temperatures (Gusek, 2002; Kuyucak et al, 2006; Nordwick et al, 2006; Alexco, 2012). Starting passive systems in warm ambient temperatures at the beginning of the summer is often the best way to ensure the survival and establishment of SRB and macrophytic communities. Burial of the organic layer can also aid in the establishment of dense biotic communities (Kuyucak et al, 2006). More specifically, when a new PTS is commissioned in the fall, establishment of the microbial community and satisfactory removal efficiencies might require 3 to 12 months (Wilmoth, 2002; Hedin, 2013). Sufficient establishment of biotic communities can be a challenge in cold climates with short growing seasons, but is often crucial to the efficient functioning of PTSs.

3.3.3. Maintenance of constant flow

Constant, steady flow through a PTS is essential to maintain hydraulic retention times and to avoid freezing. Hydraulic failure due to freezing may occur if flow is decreased below a minimum rate. Additionally, increased flow during spring run-off can be detrimental to the efficiency of a PTS. Flow through PTSs should be maintained at a constant rate through different seasons to reduce the potential of flushing out heavy metals (August et al, 2002; Nordwick, 2006). Flushing of metals may also occur as a result of system shutdowns or addition of new treatment cells (Duncan, 2010). To aid in maintaining constant flow, flow equalization ponds should be designed upstream of the treatment system. Overflows may also be necessary to accommodate spring run-off or surges in flow (Kuyucak et al, 2006). Bypasses may be necessary for components that may not function in winter. Parallel systems can also be used to accommodate increases in flows (Ettner, 2007). Recirculation of treated water at times of low flow volume may also be a solution for some PTSs. While maintaining constant flow is not an easy task in areas with extreme seasonal variation in flow rates, solutions do exist and further innovation will continue to improve this aspect of PTSs.

3.3.4. Liquid carbon sources

Bioreactors that rely on bacterial activity to treat mine-impacted water experience reductions in organic matter decomposition during the winter when cold temperatures cause limited productivity. To offset the reduction in organic matter decomposition, which feeds SRB, the addition of liquid carbon may be advantageous (Tsukamoto et al, 2004; USEPA, 2005; Sobolewski, 2010 Alexco, 2012; Gould et al, 2012). Often, the increased and year-round treatment performance more than offsets increased costs incurred by the need for reagent addition (Sobolewski, pers. Comm., October, 2014). Glucose, lactate/acetate, ethanol, methanol, ethylene glycol and other carbon sources have all been used to supplement bioreactors. Finding the optimal liquid carbon source and concentration required often relies on bench or lab-scale testing.

3.4. Knowledge gaps

Mining companies in the North attempting to implement PTSs face many challenges and despite the growing body of knowledge on their performance in cold climates, there are areas that require future research. As more cold climate PTSs are developed and implemented, our understanding of the factors affecting their efficiency and long term performance will continue to grow.

3.4.1. Seasonal variation in efficiency

While some studies have assessed seasonal variation in pilot-scale PTSs (Goulet & Pick, 2001; August et al, 2002; Gusek, 2002; Hedin, 2013; Panos et al, 2013), a greater understanding of the factors that affect efficiency losses would help PTS designers maximize the potential of their PTS. Understanding the metabolic activities of SRB and the entire scope of the microbial communities present in bioreactors and wetlands at permanently low temperatures is crucial (Robador et al, 2009). More importantly, studies are needed to determine if a relationship can be established between summer and winter metal removal rates and rates of sulfate reduction, carbon consumption, metal precipitation and sorption (Gammons et. al., 2000). Such seasonally-adjusted rates will support the development of seasonally-adjusted treatment performance and design criteria for PTSs that function year-round. Often, site-specific factors such as precipitation, temperature, pH, type of mine discharge and PTS design will interact in novel ways creating unique situations. However, with more PTS being built in cold climates, patterns about the behaviour of such systems are starting to emerge.

3.4.2. Abiotic factors in winter

A number of studies have highlighted the importance of abiotic factors in removing metals during winter months (e.g., Kwong and Whitley, 1993; MacGregor, 2002). Freeze/thaw cycles affect metal attenuation in sediments and can cause diffusion into permafrost which may lead to long-term storage (MacGregor, 2002). Sedimentation may also play an important role in CWs in the winter (Wittgren & Maehlum, 1997). These abiotic factors could be incorporated into the design of PTSs to improve winter treatment performance. As in Section 3.3.1, these factors may be unique in each setting; however as more cold climate PTSs are developed, the interplay between biotic and abiotic factors will be better understood.

3.4.3. Local species and feedstocks

Most of the research about wetland plants and organic substrates in PTSs has been done in temperate regions. As PTSs are introduced into new environments, native species and local feedstocks will have to be identified. Native plant species should be chosen to minimize the risk of introducing invasive species and to ensure that the plants are adapted to their environment. Identification of native plants that are metal tolerant and are low shoot accumulators (Stolz & Greger, 2002) requires lab, bench and pilot trials. Additionally, locally available substrates for SRB activity in bioreactors and CWs should be identified. Ideally these will be waste materials that are available on-site or nearby. While organic substrates like manure, peat, woodchips and sawdust might be easily obtained in more southern areas, these materials are often not as plentiful in remote, northern mine sites. Identification and characterization of potential feedstocks would allow PTS designers to choose the appropriate substrate for their system.

3.4.4. Expected lifetime of PTSs in cold climates

The design life of PTSs are generally not well-understood, especially in cold climates, although many natural wetlands seem to demonstrate the longevity sought after by PTS designers. Predicting the lifetime of a PTS is essential to planning its construction, as well as potential maintenance requirements. Characterization of potential organic substrates can help to optimize the long-term performance of bioreactors (Neculita et al, 2010). Research into organic substrates available in the north and their behaviour in PTSs would serve to better predict the potential lifespans of cold climate PTSs. Further research and monitoring of natural wetlands receiving mine drainage might also help elucidate the mechanisms and characteristics of long term treatment wetlands.

3.4.5. Optimal organic carbon source

Addition of a liquid carbon source to PTSs can be a useful approach to offset the loss of efficiency in cold temperatures. Reduced decomposition of organic matter can cause a reduction in microbial activity and populations. SRB and other microorganisms can be fed with a liquid carbon source to maintain their activity and population throughout the winter months. Studies have assessed various different organic carbon sources for supplementing bioreactors in cold climates (Tsukamoto et al, 2004; EPA, 2005; Gould et al, 2012; Sobolewski, 2010). Further research is required to assess the effect of carbon feed and the rate of carbon consumption in order to help predict the carbon amounts required to design efficient PTS in cold climate locations. Optimizing a carbon source for cold climate bioreactors will enhance their performance; however, the engineering requirements and costs for systems that supply organic carbon sources should also be assessed.

4. CONCLUSION

With increasing industrial activity in northern Canada there is a need for research, development and implementation of cost effective systems that ensure long term environmental protection. Passive treatment of mine impacted water has become increasingly common over the last 30 years. Passive treatment systems (PTSs) often consist of constructed wetlands (CWs), bioreactors, or some combination and/or variation on both of these components. These PTSs are sustainable alternatives to active treatment systems and represent the next generation of mine water treatment.

Natural wetlands have been treating mine-impacted water in cold climates since mining activity began and provide important information about the processes that drive PTSs, as well as providing insight into the potential for the long-term performance of CWs. Numerous studies have found evidence of metal retention in natural wetlands from both natural and anthropogenic sources of contamination. Following observed natural attenuation processes and metal retention demonstrated in natural wetlands, engineered PTSs have been designed and implemented to treat contaminated water worldwide.

Mine sites in northern environments are often located in remote locations with limited access and power, short growing seasons and cold winter temperatures, conditions which can present challenges for PTS design. These challenges however, are also often reasons for choosing PTSs, which can be constructed to operate year-round with little maintenance and no power. PTSs should ideally be able to function for decades with no power and while most systems are not truly “walk-away” scenarios, they are generally much lower maintenance than active systems. Implementation of PTSs in cold climates has been less common to date; however, several pilot scale and full scale PTSs have been built in cold climates. While the efficacy of these PTSs varies, many have successfully treated mine-impacted water to meet or exceed water quality objectives.

While success in cold climates has been variable, many PTSs have been highly successful (Norway, Bulgaria, BC, Yukon, and Quebec). Often, site-specific factors such as precipitation, temperature, pH, mine discharge chemistry and PTS design will interact in novel ways creating unique situations. As PTSs are constructed in cold climate, remote environments, patterns regarding the behaviour of such systems are starting to emerge. It is clear from this review that PTSs can be successfully implemented in cold climates, provided extensive planning and research in regards to design and construction are taken into consideration to address challenges encountered in northern environments. Some of the challenges facing PTSs in cold climates include hydraulic failure due to freezing in winter, reduced microbial activity, and increased water flows during spring melt. Many adaptations have been developed to overcome these challenges such as burial and insulation of pipes to prevent freezing, allowing sufficient time for the biotic community to establish following construction, ensuring constant flow over the season and identifying appropriate feedstocks and carbon sources that promote SRB and other microorganisms. Due to not only the site-specific challenges associated with implementing PTSs, but the additional challenges in cold climates, laboratory, bench- and field- scale experiments are needed prior to full scale implementation. However, due to the many advantages PTSs provide over active systems, these systems should be strongly considered for use in the North.

5. REFERENCES

Alexco. 2012. Galkeno 900 sulphate-reducing bioreactor 2008-2011 operations: Final Report. Prepared for Elsa reclamation and Development Corporation.

August EE, McKnight DM, Hrncir, DC & Garhart, KS. 2002. Seasonal Variability of Metals Transport through a Wetland Impacted by Mine Drainage in the Rocky Mountains. Environmental Science & Technology. 36(17): 3779-3786.

Baldwin SA & Hodaly H. 2003. Selenium uptake by a coal mine wetland sediment. Water Quality Resources Journal of Canada. 38(3): 483-497.

Banks D, Younger PL, Arnesen RT, Iversen, ER, Banks SB. 1997. Mine water chemistry: the good, the bad & the ugly. Environmental Geology 32(3): 157-174.

Batty LC & Younger PL. 2004. Growth of Phragmites australis (Cav.) Trin ex. Steudel in mine water treatment wetlands: effects of metal and nutrient uptake. Environmental Pollution 132: 85-93.

Bedessem ME, Ferro AM, Hiegel T. 2007. Pilot-scale constructed wetlands for petroleum-contaminated groundwater. Water Environment Research. 79(6): 581-586.

Beining BA & Otte ML. 1997. Retention of metals and longevity of a wetland receiving mine leachate. In: *Proceedings from the National Meeting of the American Society for Surface Mining and Reclamation*, Austin, Texas, May 10-16, 1997.

Biermann V, Lillicrap AM, Magana C, Price B, Bell RW, Oldham CE. 2014. Applicability of passive compost bioreactors for treatment of extremely acidic and saline waters in semi-arid climates. Water Research. 55: 83-94.

Blais, JF, Djedidi Z, Cheizh RB, Tyagi RD, Mercier G. 2008. Metals precipitation from effluents: Review. Practice Periodical of Hazardous, Toxic and Radioactive Waste Management. 12: 135-149.

Bless D, Park B, Nordwick S, Zaluski M, Joyce H, Hiebert R & Clavelot C. 2008. Operational lessons learned during bioreactor demonstrations for acid rock drainage treatment. Mine Water and the Environment. 27:241–250.

Blumenstein EP & Gusek JJ. 2008. Designing a biochemical reactor for selenium and thallium removal from bench scale testing through pilot construction. In: *Hydrometallurgy: Proceedings of the Sixth International Symposium, Society for Mining, Metallurgy, Exploration Inc.* Phoenix, AZ.

Boyle, R.W. 1965. Geology, Geochemistry, and Origin of the Lead-Zinc-Silver Deposits of the Keno Hill-Galena Hill area, Yukon Territory. Geological Survey of Canada Bulletin 111.

Burns BE. 2000. Investigations into passive wetlands treatment of mine drainage to remove heavy metals at various sites at UKHM. Mining and Petroleum Research Group report 2000-3, 86p.

Camenzuli D, Freidman BL, Statham TM, Mumford KA & Gore DB. 2014. On-site and in-situ remediation technologies applicable to metal-contaminated sites in Antarctica and the Arctic: a review. Polar Research. 33: 21522.

Canty, M. 2000. Innovative in situ treatment of acid mine drainage using sulfate-reducing bacteria. In: *Proceedings 5th International Conf. Acid Rock Drainage,* Denver, 2000, 1139-1147.

Contango Strategies. 2013. Minto Mine Constructed Wetland Treatment Research Program 2013 Progress Update. Prepared for Minto Explorations Ltd.

Demchak J, Morrow T & Skousen J. 2001. Treatment of acid mine drainage by four vertical flow wetlands in Pennsylvania. Geochemistry: Exploration, Environment, Analysis. 2: 0–0.

Deng H, Ye ZH, Wong MH. 2004. Accumulation of Pb, zinc, Cu and Cd by 12 wetland plant species thriving in metal-contaminated sites in China. Environmental Pollution. 132: 29-40.

Duncan WFA, Mattes AG, Gould WD, and Goodazi F. 2002. Multi-stage biological treatment system for removal of heavy metal contaminants. In: *Proceedings from Remediation Technologies Symposium.* Banff, Alberta, Canada.

Duncan WFA. 2010. Long Term Operation of Engineered Anaerobic Bioreactors and Wetland Cells Treating Zinc, Arsenic and Cadmium in Seepage – Results, Longevity, Cost and Design Issues. PhD Thesis. University of Victoria.

Eger P, Melchert G, Antonson D, Wagner J. 1991. The use of wetland treatment to remove trace metals from mine drainage at LTV's Dunka Mine. In: *Proceedings from the American Society for Surface Mining and Reclamation Meeting*. Duluth, MN.

Eger P & Eger P. 2005. Controlling mine drainage problems in Minnesota: Are all wetland treatment systems really above average? In: *Proceedings from the National Meeting of the American Society of Mining and Reclamation*. June 19-23, 2005. Published by ASMR, 3134 Montavesta Rd., Lexington, KY 40502.

Eger P & Beatty CLK. 2013. Constructed Wetland Treatment Systems for mine drainage – Can they really provide green and sustainable solutions? In: *Proceedings from the International Mine Water Association Annual Conference: Reliable Mine Water Technology.* Wolkerdorfer, Brown & Figueroa (Eds). Golden, CO.

EPA. 2000a. Wastewater Technology Factsheet. Wetlands: Free Water Surface Wetlands. EPA 832-F-00-024. Washington, DC. Available at http://water.epa.gov/infrastructure/septic/upload/free_water_surface_wetlands.pdf (Accessed on Nov. 10, 2014).

EPA. 2000b. Wastewater Technology Factsheet. Wetlands: Subsurface flow. EPA 832-F-00-023. Washington, DC. Available at www.epa.gov/owmitnet/mtbfact.htm (Accessed on Nov. 10, 2014).

EPA. 2005. SITE Technology Capsule: Compost-free bioreactor treatment of acid rock drainage. Available from http://nepis.epa.gov/Adobe/PDF/P100EXTE.pdf (Accessed on Nov. 10, 2014).

Eppley RL & Gusek JJ. 2009. Sulfate reducing bioreactor treating acidic coal mine influenced water (MIW) in Western Pennsylvania. In: *Proceedings from the Society for Mining, Metallurgy and Exploration (SME) Annual Meeting.* Phoenix, AZ.

Ettner, DC. 2007. Passive mine water treatment in Norway. IN: *MWA Symposium 2007: Water in Mining Environments,* R. Cidu & F. Frau (Eds), 27th - 31st May 2007, Cagliari, Italy.

Fortin D, Goulet R & Roy M. 2000. Seasonal cycling of Fe and S in a constructed wetland: the role of sulfate-reducing bacteria. Geomicrobiology Journal. 17(3): 221-235.

Galletti A, Verlicchi P & Ranieri E. 2010. Removal and accumulation of Cu, Ni and Zn in horizontal subsurface flow constructed wetlands: Contribution of vegetation and filling medium. Science of the Total Environment. 408: 5097–5105.

Gammons CH, Drury WJ & Li Y. 2000. Seasonal Influences on Heavy Metal Attenuation in an Anaerobic Treatment Wetland, Butte, Montana. In : *Fifth International Conference On Acid Rock Drainage*, Denver, CO. May 21-24, 2000 SME, Littleton, CO.

Germain D and Cyr J. 2003. Evaluation of biofilter performance to remove dissolved arsenic: Wood Cadillac. In: Proceedings from Mining and the Environment, Sudbury, Ontario, May 25-28, 2003.

Giaever HM. 2003. Experience and results from the northernmost constructed wetland in Norway. In: Mander U & Jenssen PD. (Eds.) Constructed wetlands for wastewater treatment in cold climates: Advances in Ecological Sciences. 11: 215-235.

Gould WD, Cameron R, Morin L, Bedard P, Lortie L. 2012. Effect of lactate/acetate and glucose amendments on low temperature performance of anaerobic bioreactors treating simulated mine drainage. In: *Proceedings of the 9th International Conference on Acid Rock Drainage.* Ottawa, Canada. May, 2012.

Goulet RR & Pick FR. 2001. Changes in dissolved and total Fe and Mn in a young constructed wetland: Implications for retention performance. Ecological Engineering. 17: 373–384.

Groudev S, Georgiev P, Spasova I & Nicolova M. 2008. Bioremediation of acid mine drainage in a uranium deposit. Hydrometallurgy. 94: 93–99.

Gusek JJ. 2002. Sulfate-reducing bioreactor design and operating issues: Is this the passive treatment technology for your mine drainage? In: *Proceedings from the National Association of Abandoned Mine Land Programs Annual Conference.* September 15-18, 2002, in Park City, Utah

Gusek JJ. 2008. Passive Treatment 101: An overview of technologies. For presentation at the U.S. EPA/National Groundwater Association's Remediation of Abandoned Mine Lands Conference, Denver, CO, October 2-3, 2008.

Gusek JJ, Buchanan RJ & Sorells D. 2013. Filtration-diverting cap and full-scale biochemical reactor operation at the Iron King/Copper Chief Mine, Arizona. In: *Proceedings from the International Mine Water Association Annual Conference: Reliable Mine Water Technology.* Wolkerdorfer, Brown & Figueroa (Eds). Golden, CO.

Gusek JJ & Conroy 2007. Hybrid treatment systems for very acidic mining influenced water. In: *Proceedings from the National Meeting of the American Society of Mining and Reclamation*. Gillette, Wyoming.

Gusek JJ, Kelsey J, Schipper R & Shipley B. 2011. Biochemical reactor construction and mine pool chemistry changes, Golinsky Mine, California. In: *Proceedings from the 2011 National Meeting of the American Society of Mining and Reclamation*, Bismarck, ND. June 11 - 16, 2011. RI. Barnhisel (Ed.) Published by ASMR.

Gusek JJ & Schneider R. 2010. Passive management of mining influenced water at the Haile Gold Mine, SC. In: *Proceedings from the 2010 National Meeting of the American Society of Mining and Reclamation*, Pittsburgh, PA. June 5 – 11, 2010. RI. Barnhisel (Ed.) Published by ASMR.

Gusek JJ & Wildeman TR. 2002. A new millennium of passive treatment of acid rock drainage: Advances in design and construction since 1988. In: *Proceedings from the National Meeting of the American Society for Mining and Reclamation*. Lexington, Kentucky.

Gusek JJ, Wildeman TR, Mann C & Murphy D. 2000. Operational results of a 1200 gpm passive bioreactor for metal mine drainage, West Fork, Missouri. In: *Proceedings from the Fifth International Conference on Acid Rock Drainage Volume II.* Published by SME.

Hambley AG. 1996. Removal of nickel from mine water by a natural wetland in Northern Manitoba. MSc. Thesis. University of Manitoba.

Hashim MA, Mukhopadhyay S, Sahu JN, Sengupta B. 2011. Remediation technologies for heavy metal contaminated groundwater. Journal of Environmental Management.

Hedin RS, Nairn RW, & Kleinmann RLP. 1994. The passive treatment of coal mine drainage. US Bureau of Mines Information Circular IC 9389. Available from http://www.hedinenv.com/pdf/ptcmd.pdf (Accessed on Nov. 10, 2014).

Hedin, RS. 2013. Temperature independent removal of iron in a passive mine water system. In: *Proceedings from the International Mine Water Association Annual Conference: Reliable Mine Water Technology.* Wolkerdorfer, Brown & Figueroa (Eds). Golden, CO.

Hiley P. 2003. Performance of wastewater treatment and nutrient removal wetlands (reedbeds) in cold temperate climates. In: Mander U & Jenssen PD. (Eds.) Constructed wetlands for wastewater treatment in cold climates: Advances in Ecological Sciences. 11: 215-235.

INAP. 2010. Global Acid Rock Drainage, GARD guide. Version 0.8. December 2010. Available at www.gardguide.com (Accessed on Nov. 10, 2014).

Jarvis AP & Younger PL. 1999. Design, construction and performance of a full-scale compost wetland for mine-spoil drainage treatment at Quaking Houses. Journal of the Chartered Institution of Water and Environmental Management. 13: 313-318.

Johnson DB & Hallberg K. 2005. Acid mine drainage: remediation options: a review. Science of the Total Environment. 388:3–14.

Kadlec RH & Reddy KR. 2001. Temperature effects in treatment wetlands. Water Environment Research. 73(5):543-556

Kadlec RH, Axler R, McCarthy B & Henneck J. 2003. Subsurface treatment wetlands in the cold climate of Minnesota. In: Mander U & Jenssen PD. (Eds.) Constructed wetlands for wastewater treatment in cold climates: Advances in Ecological Sciences. 11: 215-235.

Kepler DA & McCleary EC. 1994. Successive alkalinity-producing systems (SAPS) for the treatment of acidic mine drainage. In: *International Land Reclamation and Mine Drainage Conference*, U.S. Bureau of Mines SP 06A-94, April 24-29, 1994, Pittsburgh, PA.

Kuyucak N, Chabot F, and Martschuk J. 2006. Successful implementation and operation of a passive treatment system in an extremely cold climate, Northern Quebec, Canada. In: *Proceedings of the 7th International Conference on Acid Rock Drainage (ICARD),* March 26-30, 2006, St. Louis, MO. pp 980-992.

Kwong YTJ & Whitley WG. 1993. Heavy metal attenuation in northern drainage systems. In: *Proceedings of the 9th International Northern Research Basin Symposium Workshop,* Canada 1992, NHRI Symposium No.10. Prowse TD, Ommanney CSL and Ulmer KE (Eds), pp.305-322.

Lesage E, Rousseau DPL, Meers E, Tack FMG & DePauw N. 2007. Accumulation of metals in a horizontal subsurface flow constructed wetland treating domestic wastewater in Flanders, Belgium. Science of the Total Environment. 380: 102–115

Libero C. 2007. Étude écotoxicologique au site minier Wood Cadillac avant et après bio-traitement passif. Master thesis for the University of Montreal, 161p.

MacGregor D. 2002. Natural Attenuation of heavy metals in shallow subsurface soils over permafrost downslope of Galkeno 300 mine adit, United Keno Hill Mines, Central Yukon. Available at http://www.geology.gov.yk.ca/pdf/MPERG_2003_5.pdf (Accessed on Nov. 10, 2014).

Maine MA, Sune N, Hadad H, Sanchez G & Bonetto C. 2006. Nutrient and metal removal in a constructed wetland for wastewater treatment from a metallurgic industry. Ecological Engineering. 26: 341-347.

Mander U & Jenssen PD. 2003. Constructed wetlands for wastewater treatment in cold climates. Advances in Ecological Sciences 11. WIT Press.

Mattes A, Duncan WF & Gould WD. 2004. Biological removal of arsenic in a multi-stage engineered wetlands treating a suite of heavy metals. In: *Proceedings of the 28th Annual British Columbia Mines Reclamation Symposium*. Cranbrook, BC.

Mays PA & Edwards GS. 2001. Comparison of heavy metal accumulation in a natural wetland and constructed wetlands receiving acid mine drainage. Ecological Engineering. 16: 487–500.

MEND. 1996. Review of passive systems for treatment of acid mine drainage. MEND Report 3.14.1. 197p. May 1996. Available from http://www.epa.gov/region1/superfund/sites/elizmine/43547mend.pdf (Accessed on Nov. 10, 2014).

Mitsch WJ & Wise KM. 1998. Water quality, fate of metals, and predictive model validation of a constructed wetland treating acid mine drainage. Water Resources. 32(6):1888-1900.

Monhemius, AJ. 1977. Precipitation diagrams for metal hydroxides, sulfides, arsenates and phosphates. Transactions of the Institute of Mining and Metallurgy. 86: 202-206.

Nairn RW. 2013. Carbon dioxide impacts passive treatment effectiveness and carbon footprint. In: *Proceedings from the International Mine Water Association Annual Conference: Reliable Mine Water Technology.* Wolkerdorfer, Brown & Figueroa (Eds). Golden, CO.

Neculita C, Zagury GJ, Kulnieks V. 2007. Short-term and long term bioreactors for acid mine drainage treatment. In: *Proceedings of the Annual International Conference on Soils, Sediments, Water and Energy*: Vol 12, Article 2. Available at http://scholarworks.umass.edu/cgi/viewcontent.cgi?article=1030&context=soilsproceedings (Accessed on Nov. 10, 2014).

Noller BN, Woods PH, & Ross BJ. 1994. Case studies of wetland filtration of mine waste water in constructed and naturally occurring systems in Northern Australia. Water Science Technology. 29: 257-266.

Nordin K. 2010. Evaluation of the effectiveness of biological treatment of mine waters. Mining and Petroleum Research Group report 2010-4, 36p., Available at www.geology.gov.yk.ca/mperg (Accessed on Nov. 10, 2014).

Nordwick S, Zaluski M, Park B & Bless D. 2006. Advances in development of bioreactors applicable to the treatment of ARD. In: *Proceedings from the 7^th^ International Conference on Acid Rock Drainage (ICARD),* March 26-30, 2006, St. Louis MO.

Nyquist J & Greger M. 2009. A field study of constructed wetlands for preventing and treating acid mine drainage. Ecological Engineering. 35: 630–642.

Obarska-Pempkowiak H & Klimkowska K. 1999. Distribution of nutrients and heavy metals in a constructed wetland. Chemosphere. 39(2): 303-312.

Oberholster PJ, Cheng PH, Botha AM & Genthe B. 2014. The potential of selected macroalgal species for treatment of ARD at different pH ranges in temperate regions. Water Research. 60: 82-92.

Pahler J, Walker R, Rutkowski T. & Gusek J. 2007. Passive removal of selenium from gravel pit seepage using selenium reducing bioreactors. In : *Proceedings from the National Meeting of the American Society of Mining and Reclamation*, Gillette, WY, 30 Years of SMCRA and Beyond June 2-7, 2007. RI. Barnhisel (Ed.) Published by ASMR.

Panos NH, Gutierrez LV & Senese AA. 2013. Influence of temperature in sulphate-reducing anaerobic bacteria (SRB) development and metal removal efficiency. In: *Proceedings from the International Mine Water Association Annual Conference: Reliable Mine Water Technology.* Wolkerdorfer, Brown & Figueroa (Eds). Golden, CO.

Plewes H, Strachotta C, McBrien P & Rey L. 2009. Wetlands treatment of mine drainage at Antamina Mine, Peru. Klohn Crippen Berger. Available at http://ostrfdownload.civil.ualberta.ca/Tailings%20and%20Mine%20Waste%202009/Remediation%201/2_wetlands%20treatment%20of%20mine%20drainage%20at%20Atamina%20mine%20peru_Plewes%20et%20al.pdf (Accessed on Nov. 10, 2014).

Reinsel MA. 1998. ABC: Anoxic Biotreatment Cell for nitrogen removal from mining effluent waters. In: *Environment & Innovation in Mining and Mineral Technology*. MA. Sanchez, F. Vergara, and SH. Castro (Eds). University of Conceptiõn-Chile.

Robador A, Bruchert V & Jorgensen BB. 2009. The impact of temperature change on the activity and community composition of sulfate-reducing bacteria in arctic versus temperate marine sediments. Environmental Microbiology. (doi:10.111/j.1462-2920.2009.01896.x)

Rutkowski T, Walker R, Gusek J & Baker M. 2010. Pilot scale treatment of selenium in gravel pit seepage water using biochemical reactor technology. In: *Proceedings from 2010 National Meeting of the American Society of Mining and Reclamation,* Pittsburgh, PA, June 5 - 11, 2010.

Sagemann J, Jørgensen BB & Greeff O. 1998. Temperature dependence and rates of sulfate reduction in cold sediments of Svalbard, Arctic Ocean. Geomicrobiology Journal. 15(2): 85-100.

Schmidtova J & Baldwin SA. 2011. Correlation of bacterial communities supported by different organic material with sulfate reduction in metal-rich landfill leachate. Water Research. 45: 1115-1128.

Sheoran AS & Sheoran V. 2006. Heavy metal removal mechanism of acid mine drainage in wetlands: A critical review. Minerals Engineering. 19: 105–116

Sobolewski A, Gormely L & Kistritz RU. 1994. Copper removal from mine drainage by an experimental wetland at Bell Copper Mine, Smithers BC. Available from http://pdf.library.laurentian.ca/medb/conf/Sudbury95/GroundSurfaceWater/GSW5.PDF (Accessed on Nov. 10, 2014).

Sobolewski A. 1996a. Copper species indicate the potential of constructed wetlands for long-term treatment of mine drainage. Journal of Ecological Engineering. 6(4): 259-271.

Sobolewski A. 1996b. Development of a wetland treatment system at United Keno Hill Mines, Elsa, Yukon Territory. In: *Proceedings of the 20th Annual British Columbia Mine Reclamation Symposium* in Kamloops, BC.

Sobolewski A. 1997. The capacity of natural wetlands to ameliorate water quality: a review of case studies. In: *Proceedings of the 4th International Conference on Acid Rock Drainage*. Vancouver, B.C. May 30–June 4, 1997.

Sobolewski A. 1999. A review of processes responsible for metal removal in wetlands treating contaminated mine drainage. International Journal of Phytoremediation. 1(1): 19-51.

Sobolewski A. 2003a. Winter performance of biological treatment systems. In: *Proceedings of Second Bi- Annual Northern Latitude Mine Reclamation Workshop.* Fairbanks, AK.

Sobolewski, A. 2003b. Evaluation of metal removal in muskeg below the Silver King Mine. A Project Report submitted to: Laberge Environmental, prepared by: Microbial Technologies, Inc. Roberts Creek, BC.

Sobolewski A. 2010. Benefits of using liquid carbon sources for passive treatment systems. In: *Proceedings of International Mine Water Association Conference: Mine Water and Innovative Thinking.* Sydney, NS.

Speer S, Champagne P & Anderson B. 2012. Pilot-scale comparison of two hybrid-passive landfill leachate treatment systems operated in a cold climate. Bioresource Technology. 104: 119–126.

Stolz E & Greger M. 2002. Accumulation properties of As, Cd, Cu, Pb and Zn by four wetland plant species growing on submerged mine tailings. Environmental and Experimental Botany. 47: 271–280.

Tasse N, Isabel, D., Fontaine, R. 2003. Wood Cadillac Mine Tailings: Design of a Biofilter for Arsenic Control. In: Proceedings from Mining and the Environment, Sudbury, Ontario, May 25-28, 2003.

Taylor GJ & Crowder AA. 1983. Uptake and accumulation of heavy metals by Typha latifolia in wetlands of the Sudbury, Ontario region. Canadian Journal of Botany. 61: 63-73.

Tsukamoto TK, Killion HA & Miller GC. 2004. Column experiments for microbiological treatment of acid mine drainage: low-temperature, low-pH and matrix investigations. Water Research. 38: 1405–1418.

USEPA. 2014. Reference guide to treatment technologies for mining-influenced water. Report EPA 542-R-14-001, 94p., March 2014. http://www.clu-in.org/download/issues/mining/Reference_Guide_to_Treatment_Technologies_for_MIW.pdf (Accessed on Nov. 10, 2014).

Valente et al. 2012. Mineralogical attenuation for metallic remediation in a passive system for mine water treatment. Environmental Earth Sciences. 66:39–54

Vyzamal J & Krasa P. 2003. Distribution of Mn, Al, Cu and Zn in a constructed wetland receiving municipal sewage. Water Science and Technology. 48(5): 299-305.

Walters E, Behum PT & Lefticariu L. 2013. Sulfate reducing bioreactor dependence on organic substrates for long-term remediation of acid mine drainage: field experiments. In: *Proceedings from the International Mine Water Association Annual Conference: Reliable Mine Water Technology.* Wolkerdorfer, Brown & Figueroa (Eds). Golden, CO.

Wieder, R.K. and G.E. Lang. 1982. Modification of acid mine drainage in a freshwater wetland. In: *Proceedings of the symposium on wetlands of the unglaciated Appalachian region*. McDonald BR. (Ed), West Virginia University, Morgantown, WV, USA. p. 43-53.

Wilmoth R. 2002. Final report – Sulfate-reducing bacteria reactive wall demonstration, Report EPA/600/R-02/053, 83p., June 2002. Available from http://www.epa.gov/nrmrl/std/mwt/mtbdocs/actiiiproj12.pdf (Accessed on Nov. 10, 2014).

Wittgren HB & Maehlum T. 1997. Wastewater treatment wetlands in cold climates. Water Science and Technology. 35(5): 45–53.

Yang et al. 2006. Long-term efficiency and stability of wetlands for treating wastewater of a lead/zinc mine and the concurrent ecosystem development. Environmental Pollution. 143: 499-512

Ye ZH, Whiting SN, Lin ZQ, Lytle CM, Qian JH & Terry N. 2001. Removal and distribution of iron, manganese, cobalt, and nickel within a Pennsylvania constructed wetland treating coal combustion by-product leachate. Journal of Environmental Quality. 30:1464–1473.

Zagury GJ & Neculita C. 2007. Passive treatment of acid mine drainage in bioreactors: short review, applications and research needs. In: *Proceedings from OttawaGeo2007: the Diamond Jubilee Conference.* Ottawa, Canada.