Appendix 2.6C
Wetland Water Treatment

CLEAR COAST
CONSULTING, INC.

Memo

To: Ryan Todd

Cc: Keith Ferguson

From: André Sobolewski

Date: July 2, 2013

Re: Wetland Water Treatment for the Blackwater Project

This document presents my evaluation of wetland treatment options for the Blackwater Mine. First, I will review relevant background information to provide a context for this evaluation. Then, I will review the anticipated flow rates and chemistry at closure from the Tailings Storage Facility (TSF) and identify the constituents that are likely to require treatment. Finally, I will evaluate the potential for a passive treatment system to remove these constituents under all the conditions anticipated at the site, including during the winter.

SITE CHARACTERISTICS

The Blackwater Project is located on the Nechako Plateau, approximately 110 km south-west of Vanderhoof, in central British Columbia. The conditions at the site have some restrictions for biological processes that could provide water treatment because of winter-time low temperatures.

The elevation of the study area ranges from approximately 1700 masl on Mount Davidson to 1000 masl in the valleys. The climate is generally cold, with a mean annual temperature estimated at -0.6°C, and with minimum and maximum mean monthly temperatures of -12.9°C and 10.0°C in January and July, respectively. Winter extends from approximately mid-October to early-May, which restricts the period of growth and strong biological activity to five months. Thus, biologically-based treatment systems that stop operating during the winter are not acceptable.

The site hydrology is dominated by the spring freshet, whereby peak flows during snow melt (May and June) are approximately 10x the mean annual flows. This means that the treatment system should either be designed to treat peak flows or that water storage should be provided to attenuate peak flows during freshet. However, these peaks will be attenuated in post-closure by the storage capacity offered by the pit and the TSF.

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The overall site topography is gently hilly, with relatively flat areas in the valley bottom below the proposed TSF Dam D and adjacent to Davidson Creek. This relatively flat area is estimated to cover 70-80 hectares and could be used to construct a treatment wetland (Figure 1). In addition, wetlands are to be established on the surface of the TSF cells C and D.

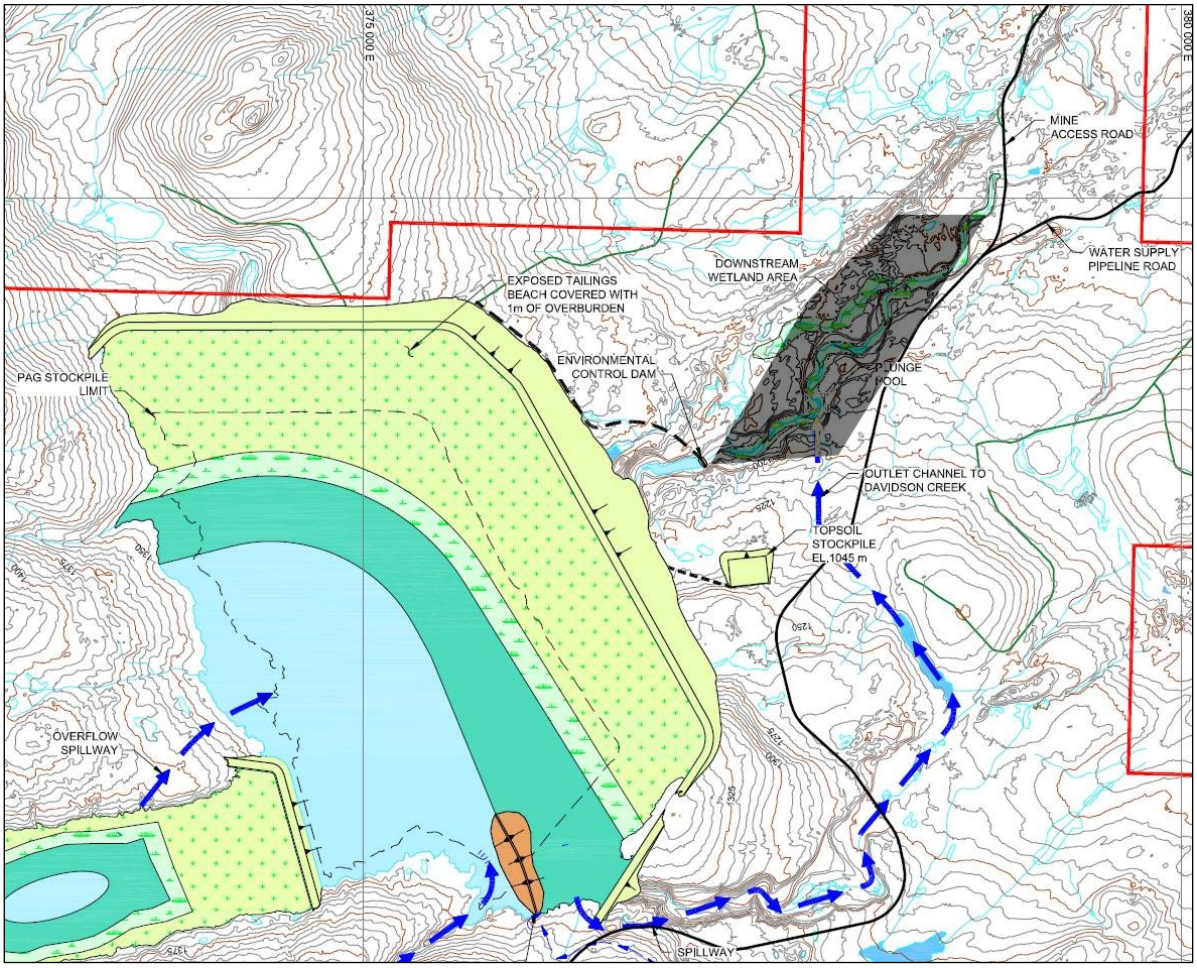

Figure 1. Gently sloping area below dam (shaded) available for a treatment wetland.

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PREDICTED MINE DRAINAGE CHEMISTRY AND ACCEPTABLE AMBIENT CONCENTRATIONS

During operations and until the pit lake overflows, tailings pond supernatant, dam runoff and seepage would be recycled to the process so that no surface water and very minimal seepage discharge (< 1 L/s) would occur to Davidson Creek based on seepage modeling conducted by Knight Piesold. The TSF will discharge via a spillway to Davidson Creek after the pit lake and TSF fill, approximately 38 years after mine start-up. This surface water discharge could either flow through a downstream constructed wetland or bypass the treatment facility as required. Wetlands would also be constructed on top of the two TSF cells at closure to treat tailings beach runoff and pit water as discussed previously.

Flow rates for the mine discharge were modeled and predicted by Knight-Piesold, based on their meteorological and hydrological investigations of the site. Further evaluations are underway.

Some seepage is predicted to flow below the dam with current estimates ranging from 10-50 L/sec at the end of mine life and into closure. TSF Embankment drain flow is currently estimated at 50 L/s. For design purposes, a seepage rate of 100 L/sec will be used. This seepage would be pumped back to the TSF during operations and while the pit lake fills. However once the pit is filled in post-closure, or sooner depending upon quality, the pumping might be discontinued so a seepage discharge rate of 100 L/s to the wetland is appropriate for design.

The water balance model identifies three distinct periods for the surface water discharge from the TSF after the pit lake and TSF begin to overflow (post-closure). During winter months, from November to April, water is predicted to flow at a rate of 60 L/sec. During spring freshet, in May and June, flows will peak at 1,300 L/sec. However, much of the flow during freshet will be uncontaminated surface runoff that can likely be diverted as clean water. For design purposes, it is assumed that flows of 200 L/sec will be directed to the passive treatment system and the rest will be diverted as clean water. Finally, the TSF will discharge water at a rate of 200-350 L/sec from July to October. For design purposes, the maximum flow that will require treatment during the summer is set at 350 L/sec. As mentioned, surface discharges from the TSF may not require any treatment depending upon its quality. TSF seepage however is expected to be contaminated, at least initially until flushed from the TSF tailings and rock porewater, and is assumed to require treatment for the purposes of this assessment.

New Gold is conducting treatment trials and modeling to predict the quality of seepage and surface water from the TSF. The chemistry of TSF supernatant at closure is expected to evolve from its chemistry during operations. The composition of TSF seepage during operations and closure may be as presented in Table 1 for key constituents, based on literature for Inco-SO2 air cyanide destruction and similarities with another gold mine. TSF supernatant might be similar quality during operations.

At the cessation of operations, the tailings beach will be covered with overburden and wetlands established on the surface of the TSF cells. TSF supernatant will be pumped to the pit for about 20 years to facilitate rapid filling of the pit. Freshwater from the upslope TSF

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catchment and water pumped from Tatelkuz Lake will replace the TSF supernatant. The combination of freshwater and reducing conditions created by the constructed wetlands is expected to result in low contaminant concentrations in the TSF supernatant by the time the pond discharges to Davidson Creek.

In the absence of reducing conditions, cadmium and zinc can remain in solution because they require higher pH to form insoluble hydroxides (Figure 2). The pH of the TSF supernatant at closure will be lower than during operations. As a worst-case, concentrations of 0.001 mg/L for Cd and 0.5 mg/L Zn are assumed for the TSF supernatant during operations.

Again as a worst case, the seepage below the TSF and embankment drainage during operations and closure is assumed to have the same composition as TSF supernatant during operations.

Table 1. Assumed metal concentrations in TSF supernatant during operations.

Parameter	**Assumed TSF Supernatant Concentrations**	
	Mean	95th Percentile
pH	7.92	8.17
Arsenic	0.009	0.013
Cadmium	0.00034	0.00081
Chromium	0.0023	0.0025
Cobalt	0.035	0.052
Copper	0.122	0.379
Iron	0.437	0.91
Mercury	0.00012	0.00012
Molybdenum	0.0258	0.040
Nickel	0.021	0.025
Selenium	0.0066	0.010
Sulphate	577	821
Zinc	0.0142	0.0453

Lorax estimated a site specific water quality objective for cadmium of 0.00006 mg/L for Davidson Creek. The 30 day guideline for zinc at low hardness is 0.0075 mg/L. The worst case drainage concentrations and acceptable concentrations in Davidson Creek for cadmium and zinc are compared in Table 2. Worst case cadmium and zinc concentrations exceed their respective guidelines by about 2 orders of magnitude respectively. If these worst case concentrations occurred, treatment of supernatant and/or seepage would be required to decrease their concentrations to acceptable levels.

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Some ammonia and nitrate will be present in TSF supernatant during operations. Ammonia will be eventually oxidized to nitrate within a few years of closure. At Equity Silver, ammonia concentrations in the tailings impoundment increased briefly after closure in 1993 (from the oxidation of cyanide residues), and then gradually decreased to <0.3 mg/L by 2009 (Figure 2, also Price and Aziz, 2012). The BC 30-Day average (chronic) concentration of total ammonia nitrogen for protection of aquatic life at pH 7 and 4 degrees C is 2 mg/L as N.

Ammonia oxidation will generate nitrate, which will be present in pond supernatant that will be transferred to the open pit at closure. However, nitrate concentrations in the pit lake will decrease over the 20 years due to algal growth, denitrification and dilution. At Equity Silver, nitrate concentrations in the TSF supernatant measured 2.1 mg/L in 2006 and peaked at approximately 20 mg/L (NO_3-N) (Figure 2). By 2012, nitrate nitrogen concentrations were below 3 mg/L (equivalent to BC freshwater aquatic life 30 day average (chronic) nitrate water quality objective).

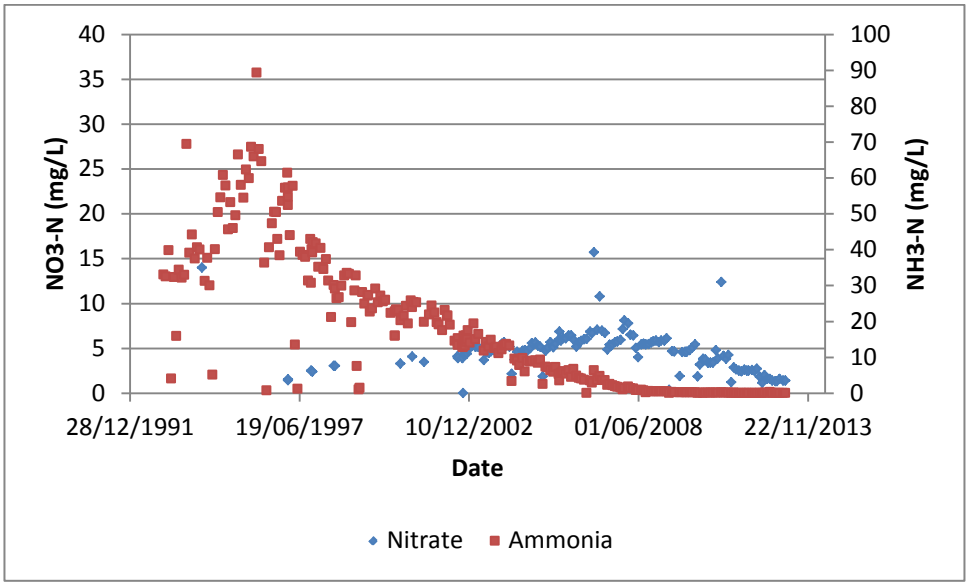

Figure 2. Ammonia and nitrate concentrations in Equity Silver tailings pond supernatant after mine closure.

A comparison of operations to post-closure (20 years later) for the Equity TSF supernatant is shown in the Table below:

Evolution of Equity TSF Supernatant from Operations to Post-Closure

Statistic	NH4		NO3		Cu		pH		Cond		SO4		Zn		Cd	
	Ops	PC	Ops	PC	Ops	PC	Ops	PC	Ops	PC	Ops	PC	Ops	PC	Ops	PC
count	16	12	NA	12	12	12	12	12	12	12	NA	12	12	12	NA	12
maximum	69.4	0.27	NA	2.74	17.10	0.038	8.27	7.42	5200	1261	NA	587	0.91	0.13	NA	0.00151
95th	50.5	0.17	NA	2.67	16.83	0.04	8.08	7.39	5096	1232	NA	584	0.62	0.11	NA	0.0012
75th	37.2	0.06	NA	2.53	11.88	0.004	7.76	7.24	4288	1172	NA	565	0.26	0.067	NA	0.0005
median	33.7	0.05	NA	2.25	3.32	0.0033	7.54	7.00	3815	1065	NA	528	0.15	0.041	NA	0.00048
25th	32.3	0.03	NA	1.62	1.64	0.0028	7.36	6.85	3738	1002	NA	464	0.11	0.028	NA	0.00036
5th	12.9	0.02	NA	1.28	1.15	0.0024	7.04	6.61	1992	848	NA	371	0.06	0.014	NA	0.00025
minimum	4.0	0.02	NA	1.14	1.14	0.0024	6.70	6.49	882	688	NA	300	0.02	0.014	NA	0.00023

units are mg/L except pH - pH units and conductivity - mohs/cm
Ops - operations (September 1992 to December 1993)
PC - post-closure (October 2012 to September 2012)

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Assuming that all the ammonia will be oxidized to nitrate, nitrate-N concentrations could reach as high as 35 mg/L shortly after closure in the pit lake. From then on, dilution can be expected to decrease its concentrations in pit water by up to two-thirds. In addition, assimilation by algal growth is expected, particularly in TSF supernatant transferred to the pit because it contains relatively elevated phosphorus levels (which usually limit algal growth)[1]. Finally, some nitrate can be biologically reduced to dinitrogen gas at the anoxic water-sediment interface, driven by organic deposition.

Accordingly, ammonia and nitrate concentrations in the pit lake and TSF supernatant after closure at Blackwater are expected to be low and meet BC water quality objectives. Assuming a worst case, nitrate-N concentrations in TSF supernatant in post-closure are assumed to be <3 mg/L and ammonia less than 0.3 mg/L.

The concentrations of ammonia and nitrate in TSF seepage after closure depend partly on their initial concentrations in supernatant. At Blackwater, nitrate concentrations in seepage are expected to be relatively low, less than 3 mg/L. However, ammonia concentrations will be somewhat elevated.

Initial concentrations of ammonia in tailings porewater might be about 30 mg/L. However, two processes will attenuate ammonia as it travels from the surface to the bottom of the TSF. First, ammonia will be retained on the surface of solids through ionic interactions. The amount of ammonia so retained depends on the nature and quantity of solids (as tailings) that can interact with ionized ammonia within the TSF. Second, ammonia can be oxidized to dinitrogen gas under anaerobic conditions through a biological process called anammox. Again, the amount of ammonia oxidized by this process is difficult to quantify.

There is evidence of ammonia attenuation inside tailings ponds of operating and closed mines. At the closed Nickel Plate mine, ammonia concentrations in TSF supernatant range from 50-60 mg/L, while its concentrations in seepage range from 0.80 – 30 mg/L, indicating an apparent attenuation rate of 50%. However, this rate is misleading because ammonia is generated in tailings due to the decomposition of thiocyanate. Thus, the attenuation rate at Nickel Plate could be substantially higher. At Musselwhite, ammonia concentrations in pond supernatant vary from 22-28 mg/L. Ammonia concentrations in seepage range from 3-6 mg/L, indicating an attenuation rate of 80-90%.

Using the above information, and assuming an attenuation rate of 80%, the post-closure concentrations of ammonia in seepage at Blackwater are conservatively predicted to be 6 mg/L.

[1] Average Total phosphorus concentrations in TSF supernatant are predicted to be 0.020 mg/L, which is moderately elevated. Lakes with Total-P concentrations of 0.010-0.030 mg/L are classified as meso-eutrophic (Wetzel, 2001), which means that neither nitrate nor phosphorus will be limiting for algal growth. The addition of phosphorus was an effective water treatment strategy at the Colomac Mine to remove thiocyanate and ammonia from tailings supernatant, as reported in: Chapman, J.T., W. Coedy, S. Schultz and M. Rkaart. 2008. Water treatment and management during the closure of the Colomac Mine. Report MV2004L8 to the NWT Water Board, April 2008.

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FEED CHEMISTRY AND DESIGN CRITERIA AT BLACKWATER

The above discussion defines the chemistry and design criteria for a treatment system at Blackwater. For the purposes of this assessment cadmium and zinc concentrations are assumed to exceed the BC MoE Guidelines, as indicated in Table 2. Ammonia concentrations also exceed the Guidelines, but nitrate concentrations are predicted to be just below the Guidelines.

Table 2. Worst case predicted TSF seepage concentrations for cadmium, nitrate, zinc and ammonia, and their acceptable concentrations in freshwater.

Constituent	Worst Case Concentration	BCWQO or Site Specific Objective
Cadmium	0.0010 mg/L	0.00006 mg/L
Nitrate (as NO_3-N)	< 3 mg/L	3 mg/L
Zinc	0.5 mg/L	0.0075 mg/L
Ammonia	6 mg/L	2 mg/L*

* SSO at 4 °C and pH 7

Three periods define the flow rates for the pit lake overflow, TSF supernatant and TSF seepage discharges. These three periods will be used in defining design flows for the passive treatment system, as indicated in Table 3.

Table 3. Design flow rates at Blackwater Project during Closure (in L/sec).

Period	Seepage	Pit + TSF Drainage	Total Flow
Winter	100 L/sec	60 L/sec	160 L/sec
Freshet	100 L/sec	200 L/sec	300 L/sec
Summer	100 L/sec	350 L/sec	450 L/sec

The flow rates during spring freshet presume that much of the runoff from pit overflow and TSF supernatant will be uncontaminated and diverted.

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PROCESS CHEMISTRY

Metal Removal

Cadmium and zinc belong to the same group in the periodic table and share a similar chemistry. Both form hydroxides, but at considerably higher pH than most metals (Figure 2). According to this figure, Fe^{3+} precipitate to low levels at acidic pH (pH 3-4), Al^{3+} and Cu^{2+} precipitate at weakly acidic pH (pH 5-6), while higher pH values are required to precipitate Fe^{2+}, Zn^{2+}, Cd^{2+} and Mn^{2+}. This is born out in practice: if only hydroxides are removed, water treatment plants must bring solution pH to 10 or more in order to precipitate cadmium and zinc.

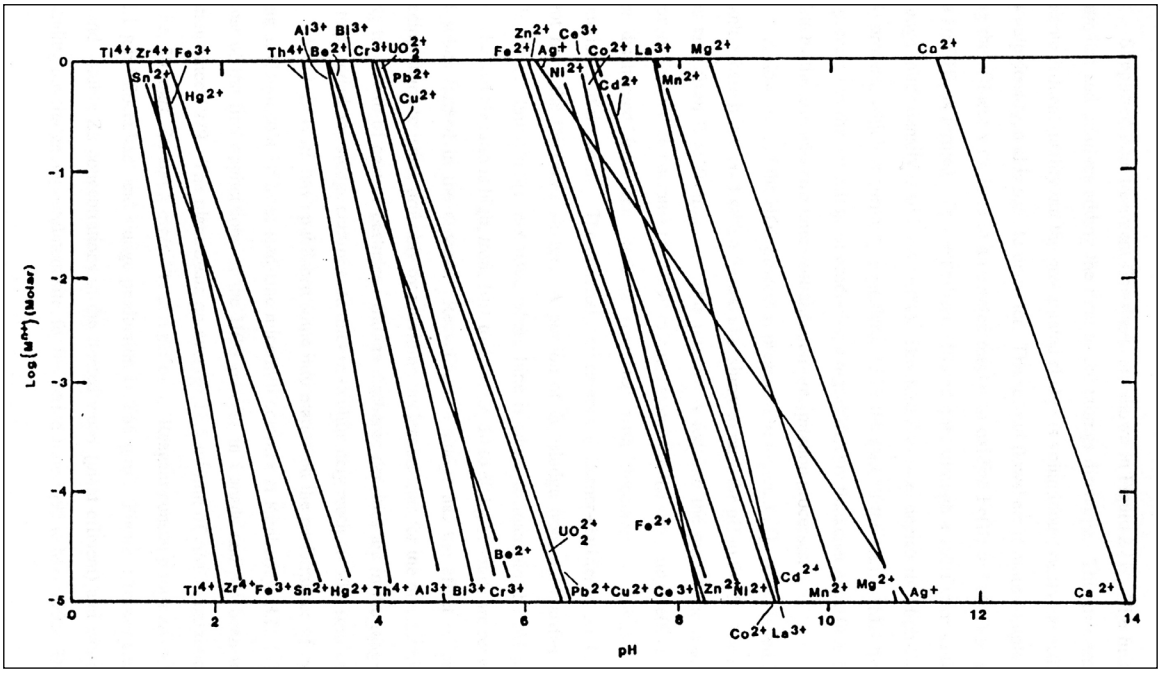

Figure 3. The hydroxide precipitation diagram at 25 ºC, from Monhemius, 1977.

Zinc readily complexes with organic matter. This may remove it from solution, as observed in systems where organic particulates bind zinc and settle below the water column. The latter was observed at Island Copper and Faro Mine, when lake fertilization promoted the formation of algal blooms that removed zinc from the water column. This may also be responsible for its removal in some of the natural systems in the Yukon, described below.

Cadmium and zinc also interact with clays and oxides of iron and manganese. Its adsorption and/or co-precipitation onto iron and manganese oxides are well-known. An example of this process is the natural attenuation of zinc from the discharge of the Galkeno 300 adit, in the Keno Hill Mining District, Yukon Territory (MacGregor, 2000). Zinc concentrations decrease from over 150 mg/L to approximately 1.5 mg/L at this site. Zinc removal was reported to be due mainly to the formation of iron and manganese oxides.

Alternatively, cadmium and zinc form highly-insoluble sulphides (Table 4). It is evident that both metal sulphides have exceedingly low solubilities; they are generally much less soluble

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than the corresponding hydroxides, and they will remain stable in reducing environments. This provides the basis for biologically-based treatment systems that use sulphate-reducing bacteria (SRB). These bacteria use sulphate to oxidize organic matter and produce bicarbonate and hydrogen sulphide as by-products. The latter reacts with metals to render them insoluble.

Table 4. Solubility products: log K_{sp} of some metal sulphides at 25 ℃.

Metal Sulphide	log K_{sp}	Metal Sulphide	Log K_{sp}
Ag_2S	-50.1	FeS	-18.1
Bi_2S_3	-100.0	HgS	-52.7
CdS	-25.8	MnS	-10.5
CoS	-21.3	NiS	-19.4
CuS	-36.1	PbS	-27.5
Cu_2S	-47.7	ZnS	-24.7

Table taken from Jackson, 1986.

Another important aspect of process chemistry to consider is that metals can only be precipitated to very low concentrations in the presence of adsorptive solid surfaces. Thus, simply adding lime to raise mine water to pH 10 will not precipitate cadmium or zinc to low concentrations. This can only be achieved by supplying abundant adsorptive surfaces and is the basis for the high-density sludge treatment process. Cadmium and zinc may react with hydrogen sulphide to form insoluble compounds, but at low concentrations, a portion will remain suspended as colloidal compounds. Their removal as sulphides from solution to very low concentrations requires that they contact adsorptive surfaces.

Any treatment system that removes cadmium and zinc to low concentrations must provide conditions that render them insoluble (high pH or high sulphide) *and* must supply abundant adsorptive surfaces.

Biological treatment options are available to remove cadmium and zinc and are usually less expensive than active (i.e. treatment plants) systems. These include treatment wetlands, bioreactors and possibly permeable reactive barriers. These systems also remove nitrate and ammonia effectively. Their applicability to the treatment of TSF drainage will be reviewed and evaluated below.

Treatment Wetlands

The capacity of wetlands to remove metals from mine drainage is well documented (Sobolewski, 1999). Wetlands are stagnant, transitional, highly-productive ecosystems that develop in water-logged or flooded, gently-sloping lands. They are characterized by the presence of hydric soils and water-tolerant vegetation (macrophytes), resulting from their periodic or continual flooding (Mitsch and Gosselink, 1993).

Both natural and constructed wetlands have been used to treat mine drainage. A wide variety of metals/metalloids have been shown to be removed from mine water, including

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aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, nickel, selenium, silver, vanadium, uranium, and zinc. They have also been used to remove non-metallic contaminants, such as ammonia, nitrate or thiosalts.

Several unique characteristics account for this unique capacity.

1. Their extraordinary productivity, reflected in the dense growth of plants, sustains a high degree of microbial diversity and activity in water, on the surface of plants, and in sediments.
2. Plant biomass is deposited in wetland sediments, and is retained as detritus in temperate and colder climates. The resulting dense network of plant stems, roots and detritus creates a large reactive surface area in contact with water, allowing biochemical transformations to proceed to completion even from dilute solutions.
3. The sluggish flow of water in wetlands allows kinetically-constrained chemical and biological reactions to proceed to completion. Their shallow depth and full exposure allows water to warm up, enhancing these reactions.
4. Lastly, the combination of intense microbial activity in wetland sediments along with the oxygen released by plant roots creates both anaerobic and aerobic zones, thereby allowing both reductive and oxidative reactions to take place simultaneously.

The design of treatment wetlands draws from these properties of wetlands to remove metals from mine drainage. Their dimensions are dictated by metal removal rates, which are determined from the scientific literature, existing comparable systems or empirically.

There are two main designs of treatment wetlands.

Surface flow wetlands direct flow of mine drainage over the surface of the wetland. In these wetlands, metal removal occurs primarily at the sediment surface, rather than in plants or in the water column. Treatment performance depends on their total surface and metal removal rates are expressed as *areal* removal rates (e.g., grams metal removed/m^2/day). Surface flow wetlands are simple to design and operate, comparatively inexpensive, but they require a substantial surface area.

Subsurface flow wetlands direct flow of mine water through an organic matrix below the sediment surface. Metal removal occurs within this matrix, fueled both by the decomposition of the organic matrix and the organic compounds released by plant roots. Treatment performance depends on their total volume and metal removal rates are expressed as *volumetric* removal rates (e.g., grams metal removed/m^3/day). Subsurface flow wetlands are more difficult to design and operate because their matrix is susceptible to plugging from the accumulation of filtered solids and deposited metals. However, they are more compact than surface flow wetlands and provide more effective treatment, particularly for low-level removal (due to their higher reactive surface area) and during winter operation.

The selection of surface vs subsurface flow wetlands depends on many factors, including feed chemistry and discharge criteria, availability of relatively flat land, inexpensive local

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source of organic material, regional climate, accessibility for maintenance, etc. All these factors will be relevant and important for the Blackwater Project.

Case Studies – Natural Wetlands

There are many examples of natural wetlands removing cadmium and zinc include a wetland downstream of an abandoned lead/zinc mine in Glendalough, County Wicklow, Ireland (Beining and Otte, 1996, reviewed in Sobolewski, 1999), a wetland below the Pacific Mine, Utah, USA (Lidstone & Anderson, 1993, reviewed in Sobolewski, 1999), and a wetland below the Woodcutter Mine, Darwin, Australia (Noller et al., 1994, reviewed in Sobolewski, 1999). These examples span the globe with regard to location, vegetation and climactic conditions, indicating that none of these constrain the ability of wetlands to remove these contaminants. Three case studies are reviewed below to provide more detail on their characteristics and performance.

Woodcutter's Mine, Australia

The discharge from the Woodcutters' Mine (a lead-zinc-silver mine located 80 km south of Darwin, N.T., Australia) has effectively been treated since 1991 by natural wetlands located downstream from the mine (See map below, taken from Noller *et al.*, 1994). The circumneutral discharge from the mine flows at somewhat less than 7,000 L/min (1850 gpm) and contains several metals (Table 3). Both cadmium and zinc are removed effectively, with cadmium concentrations decreasing from 63 to 7.8 µg/L and zinc concentrations decreasing from 6.9 to 1.7 mg/L at the wetland outlet. Moreover, annual loadings are dramatically reduced (See below). The hydraulic retention time for this system is estimated at approximately 3 days[2] (P. Woods, ERA Ranger Mine, Jabiru, NT, Australia. Personal communication).

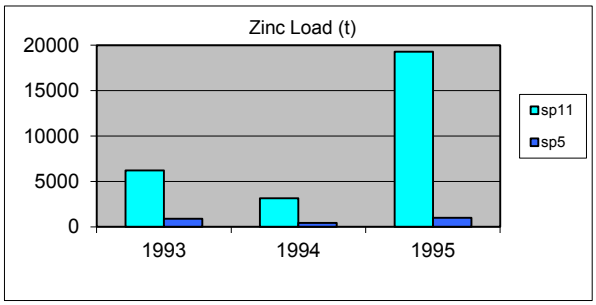

Figure 4. Zinc loads before and after natural wetland at Woodcutter's Mine.

Assuming an effective area of 100,000 m^2 (100 x 1,000 m) for the lower half of the wetland, and assuming flows of 6,600 L/min, it is possible to calculate areal removal rates for cadmium of 0.005 $g/m^2/d$ and areal removal rates for zinc of 0.49 $g/m^2/d$.

[2] This estimate is highly uncertain, due to poor information on the wetland depth and on the high, but unquantified evapotranspiration rate. However, another mine (Tom's Gully gold mine) in the same region achieves comparable metal removal rates on 2-3 day retention time (Noller *et al.*, 1994).

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Table 5. Metal concentrations in discharge from the Woodcutters Mine.

Site	As	Cd	Cu	Mn	Pb	Zn
Wetland inflow (0.0 km)	5	63	1.4	600	12	6,900
Mid-point (0.8 km)	3	63	1.7	580	7.3	5,600
Wetland discharge (2.0 km)	1	7.8	0.6	17	<0.2	1,700

Dissolved metal concentrations expressed in µg/L.

Silver Queen Mine

An adit discharges zinc-contaminated water at 10 to 100 L/min (3-25 gpm) at the former Silver Queen mine, near Houston, B.C. This discharge has enabled wetland vegetation to become established in and below the abandoned mine tailings pond. The area below the tailings pond covers 1 - 2 hectares, and has sections of open water (aided by beaver dams) as well as shallower areas vegetated with cattails (*Typha latifolia*) and sedges.

The adit discharge is slightly alkaline and contains dissolved zinc concentrations ranging from 2 to 60 mg/L. Zinc concentrations are quite high in the spring, as it is flushed by heavy rains and snowmelt. However, this initial discharge is stored in the tailings pond in the first half of the year, undoubtedly diluting zinc. Still, zinc is clearly attenuated as mine water flows through vegetated areas downstream from the adit. During the summer, its concentrations decreases from 0.5-5 mg/L in the tailings pond discharge to less than 0.1 mg/L at a discharge point below the wetland, where compliance is monitored. One sampling during the winter (March 1999) showed that zinc concentrations decreased in the wetland from 4.04 mg/L at its inlet (tailings pond discharge) to 0.33 mg/L at its outlet (Lower road culvert). Assuming a flow rate of 25 L/min and an effective treatment are of 1.5 hectare, this wetland was removing zinc at a rate of 0.009 $g/m^2/d$ during the winter.

These data indicate that the wetland that established naturally in the lowland below the Silver Queen Mine tailings pond effectively removed zinc from mine water. Moreover, treatment during the winter was only slightly less than during the summer, suggesting that a treatment wetland could be engineered for year-round treatment of mine water.

Keno Hill Mining District

There have been many reports on the natural attenuation of metals in the Keno Hill mining district. Boyle's comments (Boyle, 1965) are among the earliest recorded:

> "Numerous bogs that extract zinc from surface waters were observed in the Keno Hill area. One of these into which the mine water from the Hector-Calumet mine flows, effectively removes all of the zinc (40 ppm) in less than 2,000 feet."

Although Boyle is unclear in his reference to "bogs", referring either to wetlands or muskeg[3], observation of the area below Hector-Calumet suggests that he was probably

[3] Wetlands are flat areas which are poorly drained and remain saturated or submerged for most of the year. Muskeg is the peaty (poorly-decomposed organic matter) soil found in arctic and sub-arctic regions. Muskeg is commonly wet or

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referring to muskeg. The most remarkable – and important – aspect of this phenomenon is that zinc appears to have been consistently removed from the mid-forties to the end of production from the Hector-Calumet mine, a period of thirty years.

A natural wetland located below the Galkeno 900 adit was investigated in 1995 (Sobolewski, 1996). The wetland measured approximately 3.5 x 11.5 m. it was fed by a spring of pH 6.6 with 3 ppm zinc flowing at an estimated 0.6 L/min. No other source of input water (from the surface or subsurface) was evident. Zinc concentrations in the outflow reported at 0.27 mg/L.

Additional investigations into the mechanisms of zinc removal revealed that plant uptake was negligible, and that zinc was largely removed in the wetland sediments. This is a particular concern for local residents who shoot moose or other wildlife that would browse on the vegetation of constructed wetlands. In the sediments of this wetland, zinc was predominantly removed in association with iron and manganese oxides, with minor amounts retained in the organic, carbonate, and sulphide fractions (Table 4). In others, zinc was retained as a sulphide.

Table 3. Metal concentrations in wetland sediments and plants in the Keno Hill area[a].

	S. McQuesten* swamp	Galkeno natural swamp	Non-impacted[b] sites	Non-impacted[b] sites
Metal	Sediments/Plants *n=2*	Sediments/Plants *n=2*	Plant tissues Range	Plant tissues Mean
Cadmium	23/<0.50	66/<0.50	2.6-28	8.0
Copper	46/4.27	110/2.81	2.5-243	48
Lead	<50/4.7	98/<2.5	2.0-53	11
Zinc	1,114/132	10,345/102	26.5-1,000	143

* Control site

[a]Data expressed as mg/dry kg

[b] Ranges and means of concentrations of metals in aquatic grasses and forbs and sediments from non-impacted wetlands, as reported by Hutchinson, 1975

saturated, because of it poor drainage caused by permafrost. However, muskeg vegetation is more forest-like, whereas wetland vegetation is distinctly different, devoid of trees.

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Table 6. Concentrations of selected metal species in sediments of the Galkeno swamp[a].

	Wash	Organic	Carbonate	Fe + Mn	Sulphides	Residue	Total
Cd	<0.5[b]	<0.5	<5	15.2	1.07	0.29	17
Cu	2.1	33	36	27	27	3.9	129
Fe	11	1394	630	10637	19862	3284	35818
Mn	3	491	259	4816	128	44	5741
Pb	<1	<27	<14	8.17	7.74	6.95	23
Zn	0.99	221	116	2,532	192	18	3080

[a] Taken from Sobolewski, 1996.
[b]Data are expressed as mg/dry kg

In a study conducted in October 2001, zinc concentrations, water pH and temperature where measured in a seep below the Galkeno 900 adit. The results showed that zinc is attenuated by the muskeg and wetlands, even when water temperature is below 1 ºC. For example, zinc concentrations decreased from 0.75 ppm to 0.30 ppm as mine water flowed through a 100 meter stretch of muskeg and wetland. In another area, zinc concentrations decreased from 0.70 ppm to 0.15 ppm as mine water flowed a similar distance through muskeg and wetlands. Water temperatures at these sites ranged from -0.2 ºC to 0.2 ºC through these time periods.

Another, more thorough study was conducted in 2001/2002 at the Silver King adit, Elsa Camp. The adit discharge was followed throughout the winter as it flows onto the muskeg below. Although there was some initial glaciation, the 1.5-2.0 L/sec (25-30 gpm) flow eventually found a flow path underneath the ice and was subsequently confined to the shallow subsurface, even when temperatures decreased to –50 °C (Table 5). Samples collected along a transect downslope from Silver King showed that zinc concentrations consistently decreased from approximately 1.2 mg/L at the adit to less than 0.1 mg/L approximately 500 m below (Figure 4).

Table 7. Temperatures and zinc concentrations below Silver King adit, Elsa Camp.

Date	Temperatures	Initial Zn (ppm)	Zn (160 m)	Zn (426 m)
Dec 2, 2001	NA	1.16	0.135	
Jan 5, 2002	-40's °C	1.25	0.606	
Jan 24, 2002	-35 °C	1.14	0.396	0.071
Feb 1, 2002	-39 – -49 °C	1.1		0.026
Feb 26, 2002	-26 – -39 °C	1.25	0.327	<0.010

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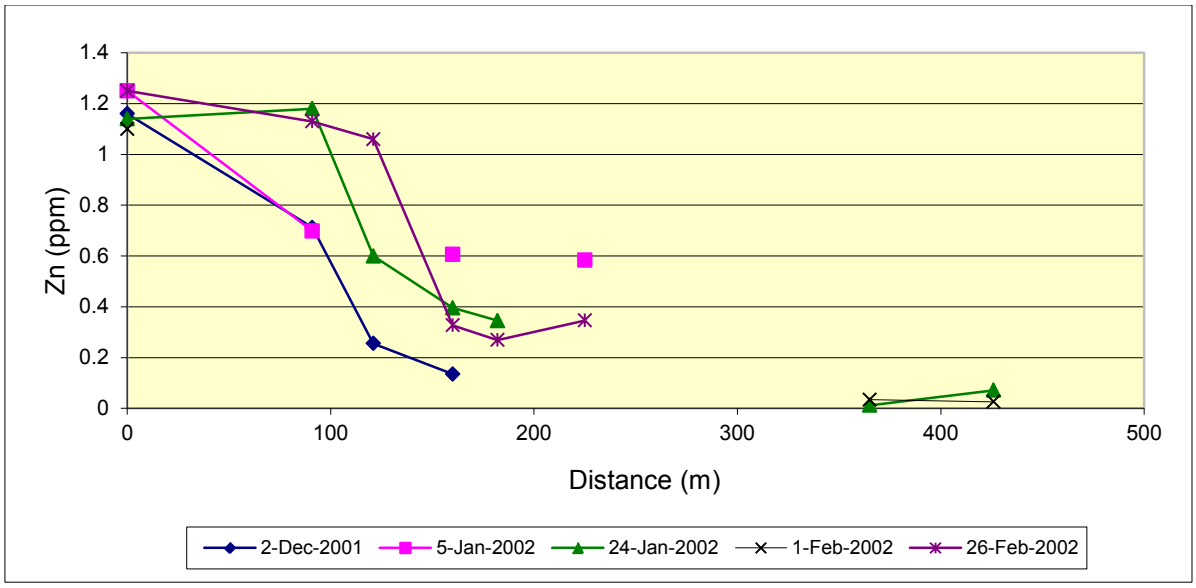

Figure 5. Zinc concentrations below Silver King adit, Elsa Camp.

These data allow estimating the areal removal rate for zinc in this wetland. Given an effective width of 30 m and a flow rate of 1 L/sec, the areal removal rate for zinc is calculated at 7.78 g/m^2/d. Although this is much higher than the areal removal rates in the other examples, it is important to note that much of this removal was due to co-precipitation with iron oxides and oxyhydroxides (which were evident during sampling).

ARCO Demonstration Wetland

An investigation of the ARCO demonstration wetland (Gammons et. al., 2000) also documented removal of cadmium and zinc on a year-round basis. The ARCO demonstration treatment wetland is located near Butte, Montana, where summers are hot (>30°C) and winter are very cold (<-20°C). Cadmium concentrations decreased in one (subsurface flow) wetland cell from 32.7 to 0.52 µg/L during the winter, whereas zinc concentrations decreased from 9,950 to 101 µg/L. The volumetric removal rates were estimated at 0.0031 g Cd/m^3/d for cadmium and 0.898 g Zn/m^3/d for zinc, assuming that half the bed volume was frozen, on average. Bacterial sulphate reduction was shown to be primarily responsible for metal removal in this wetland, with rates of sulphate reduction ranging from 0.4 mmol S/L/d to 0.05 mmol S/L/d. These rates suggest that metal removal will be 10x slower in the winter than in the summer. The authors concluded that zinc will be removed year-round in the wetland when the hydraulic retention time (HRT) is greater than 4 days.

The above results are very encouraging. They indicate that wetlands remove both cadmium and zinc effectively, even during the winter in cold climates.

Boyle's observation that:

> "Initially, this zinc is loosely bound and can be removed by acid or citrate solutions. With aging, however, the zinc partakes of the organic colloidal

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complexes and is then relatively tightly bound and unavailable to most extractants."

His observations suggest that this process of natural attenuation will be long-lived. Zinc initially removed by exchange or sorption onto iron and/or manganese oxyhydroxides/oxides (extractable by acid or citrate solutions) undergoes further geochemical transformations that render it less susceptible to remobilization.

Case Studies – Constructed Wetlands

There are few reports of constructed wetlands that remove cadmium and zinc which provide the information needed to calculate removal rates. Additionally, few investigations measure cadmium and zinc concentrations using low or sub-ppb detection limits. Typically, both metals are measured with detection limits of 0.5-2 ppb, which is insufficient for the Blackwater Project.

A small, 9 x 18.5 m wetland was investigated below the Galkeno 900 adit in Keno Hill, Yukon. The wetland was vegetated with sedges (*Carex aquatilis*) tussocks collected from a nearby donour site. Mine water taken directly from the adit was fed to the pilot wetland and fed at 18 L/min during the summer. Zinc concentrations in the inlet were fairly constant at 25 ppm and decreased to approximately 3 ppm. The areal removal rate for zinc was calculated to be 3.1 $g/m^2/day$.

A small-scale constructed wetland at the Dunka (Taconite) Mine in Minnesota received 3.8 L/min of drainage containing 0.52 mg/L zinc. The cattail-based wetland discharge contained zinc concentrations of 0.013 mg/L, effectively removing 98% of incoming zinc. The areal removal rate for this wetland was calculated to be 0.04 g $Zn/m^2/d$.

The above and other investigations of treatment wetlands rarely measure cadmium and zinc concentrations to sub-ppb detection limits. Typically, both metals are measured with detection limits of 0.5-2 ppb, which is insufficient to determine if cadmium removal is adequate for the Blackwater Project. A more recent project with a sub-surface flow wetland specifically designed to remove cadmium and zinc is relevant in this context.

A pilot-scale wetland is currently being tested in Colorado with Method Detection Limits (MDL) for cadmium of 0.000097 mg/L and MDL for zinc of 0.0016 mg/L. This sub-surface flow wetland has cattails planted over an organic matrix (mulch, manure and sulphur prills) and is constructed to operate year-round. Its operation during the winter (data for January) shows that both metals are being removed to very low levels (Figure 5). In that trial, average inflow zinc concentrations are 1.85 mg/L and average outflow concentrations are 0.0041 mg/L. For cadmium, average inflow concentrations are 0.0099 mg/L and average outflow concentrations are < 0.000097 mg/L. Hydrogen sulphide has been shown to be produced in the wetland and it is likely that these metals are removed as insoluble sulphides within the organic matrix.

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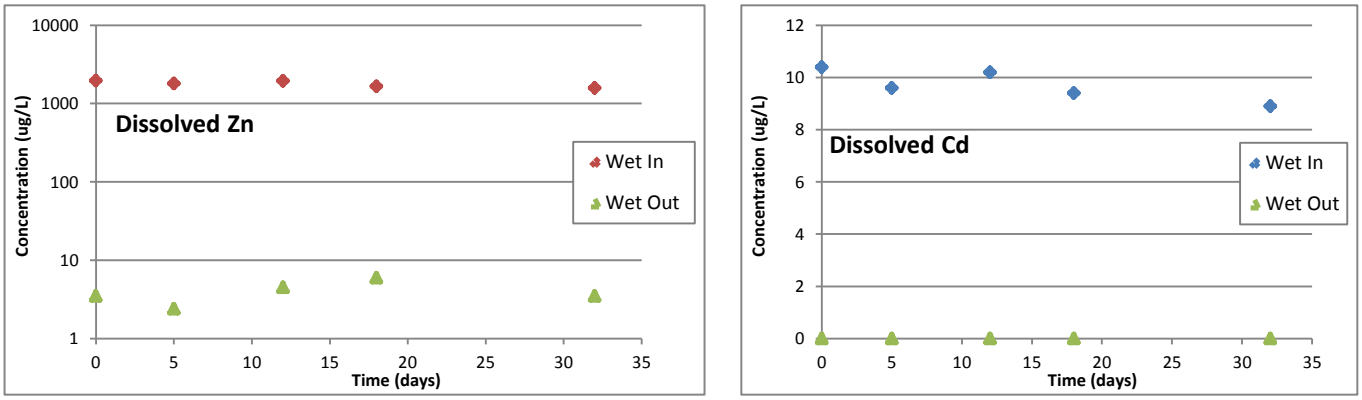

Figure 6. Influent and effluent Zinc and Cadmium concentrations in pilot-scale treatment wetland.

The volumetric removal rates for these metals are 0.00232 g/m^3/d for cadmium and 0.434 g/m^3/d for zinc. Since cadmium is removed to below detection limits in this study, it is probable that the removal rate is higher, but is constrained by input concentrations. In contrast, zinc concentrations at the wetland outlet are above detection limits, indicating that its removal rate is not so constrained.

Sizing criteria for a treatment wetland for Blackwater

The areal removal rates for various wetlands described above are tabulated below (Table 6).

Table 8. Metal removal rates for various natural and constructed wetlands.

Site	Ave input Cd (mg/L)	Areal removal rate (g Cd/m^2/d)	Ave input Zn (mg/L)	Areal removal rate (g Zn/m^2/d)
Woodcutter's Mine	0.063	0.005	6.9	0.49
Silver Queen Mine	NA	NA	4.04	0.009 (winter)
Silver King Mine	NA	NA	1.2	7.78 (winter)
Natural wetland (Yukon)	NA	NA	3.2	0.059
Constructed Wetland (Minnesota)	NA	NA	0.52	0.04
Constructed Wetland (Montana)[1]	0.0327	0.0031	9.33	0.898
Constructed Wetland (Yukon)	0.0066	0.0018	25	3.1
Constructed Wetland (Colorado)[1]	0.0099	0.0023	1.85	0.434

[1]Volumetric rates (g/m^3/day)

These removal rates vary widely, reflecting the wide range of conditions among these different wetlands. Most of the low rates reflect the fact that metals are removed to below detection limits (cadmium removal at Woodcutter's, Montana and Colorado wetlands). For that reason, the calculated rates are below the actual removal rates. The very high rate for

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zinc removal at the Silver King mine reflects the fact that it is co-precipitated with iron oxide/oxyhydroxides, with some removal due to sulphide precipitation.

Given the similar geochemistry of cadmium and zinc, the removal rates for cadmium should be similar to those for zinc. However, literature reported removal rates for cadmium are always lower than those for zinc, probably reflecting the lower input concentrations and the fact that it is often below detection limits in wetland effluents. In other words, those rates are low because cadmium is underloaded in these systems.

The most accurate removal rates during the summer (or warm weather) are those for zinc removal at Woodcutter's and at the constructed wetland in the Yukon. These rates will be used in subsequent calculations. Averaging the rates from these sites, the areal removal rate for zinc is 1.80 g $Zn/m^2/d$.

Since there is uncertainty in the rate for cadmium removal, it is set to 1/20 that for zinc, which makes it 0.090 g $Cd/m^2/d$.

Removal rates measured in the winter for wetlands vary widely. However, the volumetric removal rates for cadmium and zinc measured in Montana and Colorado are in reasonable agreement and were determined under tightly-controlled conditions. Therefore, the rates used for the calculation at Blackwater will be 0.0027 $g/m^3/d$ for cadmium and 0.67$g/m^3/d$ for zinc.

Nitrate is easily removed in treatment wetland through a combination of plant uptake and denitrification below the water-sediment interface. Published removal rates vary considerably and are almost certainly loading-constrained in most applications. Unfortunately, there are few rates published wetlands treating for mine water.

For municipal wastewater, published removal rates for surface flow wetlands range from10 kg NO_3-N/ha/d (Reddy and D'Angelos, 1997) to 20 kg NO_3-N/ha/d (Kadlec and Knight, 1996). The latter authors report that removal rates increase in direct proportion to loadings. Removal rates are even higher for subsurface flow wetlands: Fannin (2007) reported that vertical flow subsurface wetlands with 4.5m deep beds removed nitrate at 124 – 372 kg/ha/day. Assuming that removal is proportional to bed depth, this would give a removal rate of 41-124 kg/ha/day for a 1.5 m deep wetland.

Nitrate removal rates were calculated for Musselwhite, a gold mine that discharges into a natural, surface flow wetland. The mine discharged an average of 15 mg/L ammonia-N and 5.5 mg/L nitrate-N (total N equivalent to 90 mg/L NO_3). The discharge flow rates at the mine ranged from 300-600 m^3/hr (83-167 L/sec) during the ice-free season, which somewhat less than the flows predicted at Blackwater. Over 75% of inflow ammonia was converted biologically to nitrate (via nitrification) and removed from mine water. Based on influent and effluent ammonia and nitrate concentrations, the average nitrate removal rate was 350 kg/ha/day, ranging from 225-474 kg/ha/day. These rates are much higher than the ones reported above, reflecting the fact removal rates are proportional to loading rates. However, they are more representative than the other rates because the flow rates and

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chemistry of the discharge at Musselwhite resembles more closely that predicted for Blackwater.

Removal rates were also calculated as a function of temperature. Surprisingly, removal rates at 2.5 °C were only half those at 15 °C. Thus, it is expected that nitrate would be removed in the winter.

Given that nitrate concentrations at Blackwater are predicted to be lower than at Musselwhite, and assuming that removal rates are linearly proportional to concentration[4], the nitrate removal rate for a wetland at Blackwater is predicted to be 350 kg/ha/day x (90/15) = 58.3 kg/ha/day. At lower temperature, this rate will decrease by a factor of two, down to 29.2 kg/ha/day. These two rates will be used for design purposes.

Several natural and constructed wetlands remove ammonia from mines in Australia and Tasmania. Wetlands at Woodcutters (Zn-Pb-Ag) and Ranger (U) mines (Northern Territory, Australia) reportedly reduce ammonia and nitrate to below detection limits, from starting concentrations of 10's of ppm. Several wetlands in Tasmania (Hellyer Mine, Henty Gold, Beaconsfield Gold, Pasminco Rosebery Mine) are known to remove ammonia and nitrate very effectively[5]. However, removal rates (expressed as kg N removed/ha/day) have not been quantified for these wetlands or are undocumented.
Ammonia removal rates have been derived for two wetlands treating contaminated mine drainage: at the Campbell and Musselwhite mines. At Musselwhite, a wetland (one hectare) removes ammonia from approximately two million m^3 of settling pond decant during the ice-free season. Ammonia concentrations in the influent (pH 8) range from 15-20 mg/L and < 10 mg/L in the wetland effluent (pH 6.7-6.9). Areal removal rates appear to be first order in relation to temperature (from 0.5 to 20 °C) and range from 10.4 – 24.4 kg/ha/d. At the Campbell Mine, ammonia concentrations of 5-10 mg/L are decreased to 0 – 4 mg/L in a 9.8 hectare wetland, on average annual flows of 2.5 million m^3. Areal removal rates are also temperature-dependent and range from 1.5 – 15 kg/ha/d. The removal rates that will be used to calculate the areal requirement for a treatment wetland are 1.5 and 15 kg/ha/d.

Bioreactors

Like wetlands, bioreactors use biological processes to treat mine water. These are carried out by bacteria grown on an organic matrix that is contained in excavations like ponds[6]. They are operated anaerobically, promoting anaerobic bacterial processes like nitrate-, selenium- and/or sulphate-reduction[7]. Mine water is treated as it flows through the bioreactor, just as for wetlands.

[4] Ammonia removal rates are linearly proportional to incoming concentrations and loadings.

[5] Philipa Varris, Henty Gold; Wojciech Grun, Mineral Resources Tasmania (MRT), Department of Infrastructure, Energy and Resources, Tasmania. Personal communication.

[6] Bioreactors used in treatment plants are disregarded in this review, since they are not passive treatment systems.

[7] A corollary is that oxidative treatment processes, like ammonia oxidation, do not take place in bioreactors.

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In some respects, bioreactors resemble subsurface flow wetlands, but they rely entirely on the organic matter contained inside it because they do not produce organic carbon. The organic matter (e.g., manure, compost, wood mulch, etc) must either be replenished or it must be supplied externally. For this reason, bioreactors have a fixed design life (typically 25 years) before they must be replenished, whereas wetlands can treat mine water for hundreds of years.

Bioreactors have been used for the past 20 years to treat water at closed or abandoned mines, such as the ARD from East Appalachia coal mines (e.g., Kepler and McCleary, 1994), from abandoned underground mines (e.g., Canty, 2000), or neutral drainage from closed mines (e.g., Cellan *et. al*., 1997). There have been fewer bioreactors used in Canada. Two of the better-known examples are the SRB-based bioreactors at Teck's Sullivan Mine (Duncan *et. al*., 2008) and at the Tulsequah Chief Mine (Chandler *et. al*., 2010), which have been operated year-round for several years in BC.

The design of bioreactors treating mine drainage is still in its infancy, but some design guidelines have been developed over the years. The current "rule-of-thumb" to determine their dimensions gives the following: volumes are calculated from removal rates of 0.3 mol $[Me^{+2}]m^3$/day, based on input total divalent metal concentration; and surface area is calculated as 20 m^2/L/min at pH<5 based on input flow rate.

These design criteria are not universally accepted and must be modified to account for climate. In addition, their dimensions should be modified to account for the mass of metal retained/stored within the system, in order to prevent plugging issues. However, these rules-of-thumb provide a starting point for designing systems when there are limited available data. Typically, a pilot-scale facility will be constructed to treat the contaminated mine drainage to determine volumetric removal rates, seasonal variability, confirm the validity of the matrix composition (in terms of performance, hydraulic characteristics, etc), and system reliability.

Bioreactors can treat water year-round. In this regard, they are entirely comparable to the subsurface flow wetlands discussed above. However, there has been a trend in recent years to supply a liquid source of organic carbon to enhance their performance, particularly during the winter (Sobolewski, 2010). It is known that certain bacterial processes, like nitrification, completely shut down at low temperatures, whereas others, like selenium- or sulphate-reduction, are largely unaffected.

Contaminants being removed are retained inside bioreactors as well as wetlands. A key difference is that contaminants are typically deposited on the sediment surface in wetlands (free surface flow), where they will not impair flow, whereas their deposition and accumulation within the organic matrix of bioreactors can eventually impair bioreactor function from plugging problems. Thus, bioreactors are not favoured for passive treatment of mine water that contains high (> 50 mg/L) metal concentrations, unless their design involves some flushing or backwash components (e.g., Hedin Environmental, 2008). Unfortunately, self-flushing bioreactors have met with limited success and plugging remains the biggest obstacle to the use of bioreactor in treating mine drainage.

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The organic matrix inside bioreactors is a key component in their design. Typically, it comprises a mixture of inert material (e.g., river rock, limestone, etc), organic material (e.g., wood chips, mulch, compost, manure, etc), and possibly constituents that support microbial growth and activity (e.g., fertilizer, sulphur prills, etc). The proportion of each constituent in this mix aims to maximize biological activity (through high surface area) while maintaining good flow characteristics and avoiding plugging problems (through high porosity). Some bioreactors, such as the SRB-based bioreactor at Leviathan (US EPA, 2006) use completely inorganic materials for the matrix and rely on external organic carbon, supplied as ethanol, ethylene glycol, methanol or molasses. These bioreactors tend to avoid plugging problems, but they function more like treatment plants than passive systems.

Case Studies

The bioreactor operated at Teck's Trail smelter facility is one of the most thoroughly investigated full-scale systems in North America. The system originally started in 1996 as a horizontal flow wetland, but two anaerobic bioreactor cells were added upstream of the wetland in 1998 and 2000, with a holding pond constructed downstream of the entire system. The first bioreactor was taken apart and reconstructed in 2002 to address performance issues.

These many iterations in the system design were due to the harsh nature of the water being treated. The system was treating seepage from a landfill that initially contained elevated concentrations of arsenic, cadmium and zinc, as indicated in Table 7.

Table 9. Characteristics of landfill seepage at Teck's Trail smelter.

Total Flow	Arsenic	Cadmium	Lead	Zinc
(L/day)	(mg/L)	(mg/L)	(mg/L)	(mg/L)
~77,000	6.0	4.6	0.056	434

The two bioreactors were vertical-upflow cells receiving 15,000-20,000 L/day. The first bioreactor measures 25 x 18 x 3.5 m and contains a mix of 5% manure, 35% sand and 60% biosolid residues from the Celgar pulp mill). This experienced problems in the first few years of operation, due to internal erosion, slumping and deposition of sulphides on the piping system, resulting in poor performance and a number of failures. The bioreactor was repaired in 1999 and reconstructed entirely in 2002, increasing its depth to 3-7 m, adding limestone to the matrix and changing its composition to the same as the second bioreactor (see below).

The second bioreactor was constructed in 2000. The cell is built as a trapezoid: it measures 9 m wide at the top end and expands to 22 m at the down slope end. The length of the cell is 25 m for a total surface area of approximately 400 m^2. The depth changes from 2.8 m at the top end to 4.2 m at the bottom end for a total volume of approximately 1000 m^3 and a treatment volume of 800 m^3 as the biological substrate was filled to 1 to 1.5 m from the top. The excavated cell is lined top and bottom with limestone and is filled with Celgar biosolids (65%) and a 50:50 mix of sand and limestone.

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Initial zinc removal in the first bioreactor was poor (15-35%), indicating that the bioreactor was undersized for the high incoming concentrations. Arsenic and cadmium removal ranged from 70-90%, which was more acceptable. After the bioreactor was re-built, zinc removal increased to 50-60% during the summer and 30-50% during the winter. Arsenic and cadmium removal remained high during the summer (>90%), but decreased to 80-90% during the winter. When the two bioreactors were operated in series, summer removal rates for all metals reached 80-95%, whereas winter removal rates ranged from 50-90% (Table 8). Week-to-week variability in removal rates was observed and related to precipitation and temperature events.

Table 10. Average Metal concentrations in Teck bioreactors.

		Inflow	Cell #1 Outflow	Cell #2 Outflow
1999-2002				
Summer	Arsenic	31.7	1.96	0.63
	Cadmium	3.63	0.494	0.063
	Zinc	247	124	53.2
Winter	Arsenic	75.1	14.3	5.79
	Cadmium	2.84	1.05	0.446
	Zinc	230	152	129
2002-2007				
Year-round	Arsenic	36	8.3	2.7
	Cadmium	2.32	0.13	0.05
	Zinc	166	43	30.6

All values for dissolved metal concentrations, in mg/L.

Volumetric removal rates were calculated for each contaminant. These rates were based on the varying flow rates experienced at the site and included:

- Arsenic removal rates ranged from 0.22-0.36 g/m^3/day
- Cadmium removal rates ranged from 0.015-0.025 g/m^3/day
- Zinc removal rates ranged from 0.90-1.50 g/m^3/day

The zinc volumetric removal rates are similar to those presented in Table 6 for the Montana and Colorado treatment systems, whereas those for cadmium are substantially higher. However, it is worth noting that cadmium concentrations in the latter systems were typically below detection limits, whereas they did not in the above bioreactors, and are therefore more likely to be more relevant because they are not measured from loading-limited systems.

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Metals retained within the first bioreactor were characterized when it was torn down and reconstructed. During the 729 days that it was operated, the bioreactor retained 177 kg As, 24.5 kg Cd, and 895 kg Zn. Microbial and mineralogical analyses confirmed that metals were retained as sulphides within the bioreactor, particularly iron and zinc sulphides. Adsorption onto organic matter was also noted. An analysis of the organic material, particularly Total Organic Carbon, in the deconstructed bioreactor led to a predicted cell life of 21 years (range 14-34 years).

This case study highlights both the potential benefit and problems of bioreactors. The bioreactors were able to remove very high inflow metal concentrations from contaminated seepage. However, this resulted in problems with plugging and required repeated maintenance, and in one case, rebuilding, of the bioreactors.

Permeable Reactive Barriers

Permeable Reactive Barriers are structures excavated in the flow path of contaminated groundwater that are designed to remove contaminants. There are two principle types of PRBs, based on the composition of their reactive matrix: those that use elemental iron (zero-valence iron, or ZVI) and those that use an organic matrix that supports biological sulphate reduction. The latter type closely resembles SRB-based bioreactors, with the key difference that PRBs receive smaller water fluxes, and hence provide considerably longer Hydraulic Retention Times. This allows them to remove contaminants to very low levels, including sulphate.

The design of SRB-based PRBs is comparable with that of SRB-based bioreactors and will not be discussed further.

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BLACKWATER WETLAND TREATMENT SYSTEM

Surface area requirements

Although both bioreactors and treatment wetlands can remove cadmium and zinc year-round at this site, wetlands are favoured by virtue of being self-sustaining and long-lived. A subsurface flow treatment wetland is expected to remove contaminants from TSF supernatant and seepage to concentrations that are protective of freshwater.

The wetland must be large enough to accommodate the different flows and loadings that occur at different times of the year, accounting for seasonal changes in removal rates. This is done by dividing contaminant loads that must be removed by the seasonal removal rates to obtain the surface area and/or volume needed to provide full treatment.

Seasonal metal loads to be removed are presented in Table 11.

Table 11. Seasonal metal loads to be removed from combined TSF supernatant and seepage.

Contaminant	Winter Loads	Summer Loads
Cadmium	13.8 g/day	38.9 g/day
Zinc	6,912 g/day	19,440 g/day

Design volumetric removal rates are presented in Table 12 for a subsurface flow wetland. These rates are combined from volumetric rates for the subsurface flow wetlands and bioreactors reviewed above.

Table 12. Seasonal volumetric removal rates for cadmium and zinc.

Contaminant	Winter Rate	Summer Rate
Cadmium	0.0027 g/m^3/day	0.025 g/m^3/day
Zinc	0.67 g/m^3/day	1.50 g/m^3/day

Applying seasonal removal rates for these contaminants gives the surface area required for treatment, assuming a bed depth of 1.25 m. Removing 0.001 mg/L cadmium during the winter requires a volume of 5,111 m^3 and an area of 0.41 hectare (Table 13).

The recommendation by Gammon that a 4-day retention time is required to remove cadmium and zinc during the winter was used as an alternative method to size the treatment wetland for cadmium and zinc removal. For flows of 160 L/sec, a nominal 4 day HRT will require a bed volume of 55,296 m^3 and an area of 4.42 hectare. This area is ten times more the area calculated from published removal rates and is based on conservative assumptions.

While both ammonia and nitrate will be elevated during operation and immediately after closure, their concentrations will decrease in the pit lake due to dilution and attenuation and are expected to meet Guidelines in TSF supernatant. Nitrate concentrations in seepage are

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also expected to meet Guidelines. Only ammonia concentrations in seepage are expected to exceed Guidelines post-closure and require treatment.

Seasonal ammonia loads to be removed are presented in Table 13.

Table 13. Seasonal ammonia loads to be removed from combined TSF supernatant and seepage.

Contaminant	Winter Loads	Summer Loads
Ammonia	55.30 kg/day	155.52 g/day

Design areal removal rates for ammonia are presented in Table 14 for the surface flow wetlands, using the most conservative values from the Campbell and Musselwhite wetlands.

Table 14. Seasonal volumetric removal rates for ammonia.

Contaminant	Winter Rate	Summer Rate
Ammonia	1.5 kg/ha/d	15 kg/ha/d

Applying seasonal removal rates for these contaminants gives the surface area required for treatment. Removing 4 mg/L ammonia during the winter requires a surface area of 36.9 hectares and an area of 10.4 hectares during the summer (Table 13).

Table 15. Calculated surface area required to treat TSF supernatant and seepage.

Contaminant	Season	Load	Removal Rate	Volume	Surface Area
Ammonia	Winter	55.30 kg/d	1.5 kg/ha/day		36.9 ha
	Summer	155.52 kg/d	15 kg/ha/day		10.4 ha
Cadmium	Winter	13.8 g/d	0.0027 g/m^3/day	5,111 m^3	0.41 ha
	Summer	38.9 g/d	0.025 g/m^3/day	15,099 m^3	0.12 ha
Zinc	Winter	6.9 kg/d	0.67 g/m^3/day	10,316 m^3	0.83 ha
	Summer	19.4 kg/d	1.50 g/m^3/day	12,960 m^3	1.0 ha

The volumes and surface area shown in Table 15 indicate that treatment of ammonia drives the wetland design. Taken together, **a combination of 1.0 hectare subsurface flow wetland and 35 hectare surface flow wetland will remove all the metals and ammonia present in the discharge of the TSF**. These dimensions are based on worst case predictions for ammonia, cadmium, and zinc concentrations in mine drainage after closure and conservative contaminant removal rates reviewed above. As such, these are the maximum surface areas that will be required for post-closure treatment of mine drainage.

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Wetland areas, volumes and cost estimate

Figure 1 shows that there is an extended strip of land between the mine access road and the main channel below the dam that provides more than 40 hectares for surface flow and subsurface flow wetlands. These wetlands will capture and treat all the mine drainage below the dam although high flows during freshet would bypass the system as discussed previously.

The subsurface flow wetland will first receive mine water. It will be comprised of several cells with a nominal bed depth of 1.25 m and covering a surface area of 1.0 hectares. The bed matrix will be a mix of 60% limestone, 35% wood chips and 5% manure, occupying a total volume of 55,000 m^3. A field trial will be required to determine specific design aspects, including configuration that allows metal removal during winter months.

The capital cost (+/- 50%) to construct such a system is estimated based on a cost estimate (+/- 20%) for a comparable passive treatment system (surface wetland + limestone drain) to treat leachate at the closed Eurocan pulp mill in Kitimat (2011). It is assumed that the earthworks will be done by mine personnel (not a contractor), as part of their closure activities. This includes clearing, development of access and laydown areas, excavation of wetland beds, placement of material, water distribution systems, and planting for the surface and subsurface wetland. Additionally, borrow material (e.g., clay or glacial till) may be needed for construction of low permeability berms and cell bottoms, if no suitable material is available at the site[8]. Design and engineering will be assumed to be contracted.

The cost for the earthworks, if contracted, is estimated at $0.8 million. The cost for engineering, water conveyance, planting and other ancillary work for both wetlands is estimated at $0.32 million. Thus, the total labour cost is estimated at a maximum of $1.12 million.

The material costs are estimated below based on the following volumes:

- Limestone: 9,000 m^3 (15,750 tons)
- Wood chips: 5,250 m^3
- Manure: 750 m^3

Limestone costs will be very dependent on the proximity of a quarry that can supply such a volume of material. It is estimated that limestone supplied to the site will cost $70/ton[9]. The total cost for this volume of limestone will be $1.10 million.

It may be possible to substitute another type of rock for the limestone, which could reduce this cost significantly. Testwork should be done to determine if this is possible.

[8] This material may be the same as used for construction of the impoundment. It is extremely unlikely that a synthetic liner will be required to make the wetland bottom impermeable.

[9] The cost is broken down as: $15/ton for material, $35/ton for transport to mine, and $20/ton for handling.

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Wood chips should be readily available from mills in the region. The cost for obtaining the wood chips on site is estimated at $\$65/m^3$, resulting in a total estimated cost of $0.34 million.

Manure should also be available in the region, although it might be difficult to source the large quantity required for the subsurface wetland. Assuming a cost of $\$60/m^3$, the resulting cost is estimated at $0.045 million.

Thus, the total material cost for the subsurface wetland is estimated at $1.5 million. This estimate assumes that much of the labour is contracted. Thus, the total estimated cost for the subsurface wetland is $2.6 million (in 2011 CDN$).

The design life for the subsurface wetland is 25-30 years, based on estimates from comparable systems (Gammons *et. al.*, 2000; Duncan, 2010). At that time, the organic matter in the matrix may need to be replenished.

The surface flow wetlands will follow the subsurface flow wetland have a surface area of 36 hectares. It will be constructed below the subsurface wetland in the area shown in Figure 1. The wetland will comprise multiple, interconnected shallow cells through which mine water will flow by gravity. It will be vegetated with plants that grow in the area, such as cattails and sedges.

Constructing a wetland on such a scale will require some planning. Field trials will be needed to determine operating depth and other parameters that allow operation during summer and winter months. Additional field studies will also be required to determine the best location, considering local topography, soils, etc.

Construction costs are difficult to estimate because the volume of earth to be moved cannot be determined with the present information. Preparing the grounds could cost $0.54 million, based on subgrade preparation of 360,000 m^2 @ $\$1.5/m^2$. There could be $3-5 million in additional earthworks, depending on the amount of excavation and fill required to develop the wetland cells and allow water to flow by gravity, as well as development of access. Finally, there will be additional costs for infrastructure (e.g., water conveyance), planting, and engineering totaling as much as $0.7 million. Altogether, development of the surface wetland may cost between $4 and $6 million. These costs assume that mine staff and equipment will do as much of the work as possible, as part of the mine closure plan.

The above designs are based on the flows and contaminant loadings shown in Table 3 and Table 13. Flows and concentrations of seepage, embankment drainage and surface discharge at closure are still being evaluated. The greatest uncertainty with an impact on wetland size is the final ammonia concentrations. It is likely that some information on its concentrations in seepage will be available at closure, allowing us to refine the final surface area required for treatment.

Annual operating costs are estimated to range from $50-100,000/year. These costs reflect monitoring and maintenance requirements to repair damages from wildlife, frost and high water flows. Higher costs will be incurred at approximately 25 year intervals in the subsurface wetland to refurbish the organic matter due to compaction or depletion.

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ANTICIPATED TESTWORK AND PATH FORWARD

After confirming that drainage will require treatment after closure, the Blackwater mine should undertake a number of studies to properly size and design the treatment wetlands below the dam.

The contaminant loadings need to be defined more precisely. Seasonal flow rates and contaminant concentrations need to be confirmed for seepage from the embankment and below the dam. This information will update existing predictions and refine the treatment requirements at closure.

Seasonal removal rates need to be determined more precisely. Two pilot-scale wetland (surface and subsurface flow wetlands) will be required to determine these rates. In addition, the wetlands could provide information on operational issues, such as the fate of metals, the hydraulic properties of the organic matrix, etc. These wetlands should be sufficiently large (e.g., 300-500 m^2) and mature (e.g., after good vegetation is established, which will take 1-2 years) to provide information that can be used to design a full-scale system. The pilot wetlands will need to be operated for at least one year with mine drainage with a chemistry that closely resembles that predicted at closure.

The organic matrix of the subsurface flow wetland is one of the large cost items. It would be beneficial to determine if alternative, less costly materials (e.g., granitic or waste rock) could be used instead of limestone. This could best be determined by doing side-by-side comparisons between two small-scale wetlands.

Once the design criteria and wetland surface area/volume have been refined, the mine should identify areas where it/they can be located. Although the design will remain at a conceptual stage during mine operation, mine staff can survey potential sites, characterize flow patterns at these locations, identify potential challenges, such as characteristics of local soils, potential interactions with groundwater, availability of suitable construction material, etc. These tasks will help to refine the estimated capital costs for the wetlands and may affect the amount of security required by regulators.

Another important issue to resolve is the management of flows during freshet after closure. This will require a better understanding of flow patterns around the mine site at that time and development of measures to divert clean water away from the treatment wetlands.

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FATE OF MERCURY AND SELENIUM IN WETLAND SEDIMENTS

Both mercury and selenium will be retained in the treatment wetland. They will remain immobilized in the sediments, so long as they remain anaerobic. There is a possibility that they could be remobilized from the surface flow wetland, but not from the subsurface flow wetland, which will remain permanently anaerobic. Thus, any constituent released by the surface flow wetland will be captured by the subsurface flow wetland located downstream. Moreover, ingestion of mercury is prevented in the subsurface wetland, insuring that it cannot enter into the food chain.

There is no concern about the possible formation of methyl-mercury because sulphate concentrations in mine water are expected to be above the critical 250 mg/L concentrations where mercury methylation occurs. Report in the literature indicates that mercury accumulates in wetland sediments, and predicts that it will form insoluble sulphides in wetlands receiving water with high sulphate concentrations (Root, 1996). Reports on mercury methylation in wetland sediment indicate that it is carried out by sulphate reducing bacteria (SRB) in freshwater wetlands receiving water with low sulphate concentrations (Schlesinger, 1991). This is because a metabolic switch is activated in SRB at low sulphate concentrations, causing the production of an enzyme that accepts mercury as a substrate for methylation. This enzyme is not produced at sulphate concentrations above 250 mg/L, and mercury methylation becomes impossible above these concentrations.

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CONCLUSIONS

A summary of influent and effluent Cd and Zn data for the case studies discussed above and from other sources is presented below:

Summary of Wetlands and PRB Treatment System Performance

Wetland/PRB	Influent Cd ug/L	Effluent Cd ug/L	Influent Zn ug/L	Effluent Zn ug/L	Information Source
Natural Wetlands					
Woodcutters, Australia	63	7.8	6900	1700	Clear Coast Consulting
Silver Queen, BC					
summer	-	-	500-5000	<100	Clear Coast Consulting
winter	-	-	4040	330	Clear Coast Consulting
Keno Hill, Yukon					
	-	-	3000	270	Clear Coast Consulting
	-	-	1100-1250	26-71	Clear Coast Consulting
NCC	6.8	0.9	-	-	unpublished data
HSW	0.1	<0.05	-	-	unpublished data
Galkano	592	<6	168000	1550	1
Constructed Wetlands					
ARCO, Montana	32.7	0.52	9950	101	Clear Coast Consulting
Keno Hill, Yukon	6.6	-	25000	3000	Clear Coast Consulting
Dunka, Minnesota	-	-	520	13	Clear Coast Consulting
Colorado	9.9	<0.097	1850	4.1	Clear Coast Consulting
Bioreactor Trial					
Trail, BC	2320	50	166000	30600	Clear Coast Consulting
PRB (Lorax)					
Vancouver, BC	15.3	0.2	2000	<1	Lorax Environmental Consulting
Charleston, SC, USA	320	<1	1060000	<230	Lorax Environmental Consulting
VFRB					
Cadillac, QC	-	-	1350	12	2

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1 MacGregor, D., "Natural Attenuation of Zinc Occurring in Shallow Soils Downslope of Galkeno 300 Adit, Keno Hill Mining District, Central Yukon Territory, and the Implications of Residual Zinc Accumulation"

2 Kuyucak, N, F. Chabot and J. Martschuk. 2006. "Successful Implementation and Operation of a Passive Treatment System in an Extremely Cold Climate, Northern Quebec, Canada. Paper presented at 2006, 7th ICARD, March 26-30, 2006, St. Louis, MO.

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Effluent quality was variable reflecting natural vs constructed designs, higher detection limits and that systems were generally not designed for zinc and cadmium removal. However, low zinc and cadmium effluent concentrations have been achieved. The above review of the literature and case studies supports the use of wetlands to treat mine drainage at closure, for the flows and metal concentrations predicted at that time. The key challenges that this represents – treatment to very low metal concentrations, treatment of high flows, and treatment during the winter – have been shown to be met by wetlands operated under comparable circumstances.

The surface area required for year-round treatment of the predicted flows is approximately 37 hectares, an area that can be accommodated downstream of the Blackwater TSF (Figure 1). Both a surface flow and a subsurface flow wetland would be needed to remove the large ammonia loads predicted at closure, as well as the cadmium and zinc that could be present in TSF seepage. Given that these ammonia loads represent a worst-case prediction and given that there is a large degree of uncertainty in that prediction, it is possible that only half of the predicted wetland surface area would be necessary for its removal.

Information in the literature indicates that cadmium and zinc removed by these wetlands would be retained stably within their sediments. In addition, these metals would not be taken up by the plants and would not enter the food chain. Taken together, this information indicates that the proposed treatment wetlands would protect both the aquatic and terrestrial ecosystems downstream from the Blackwater TSF from metal presents in seepage.

The total cost for constructing the downstream TSF wetlands is estimated to be at most $8.6 million (2011 $CDN). Several cost savings could be realized by finding suitable, less expensive materials or using mine personnel and equipment instead of contractors. In addition, the size of the treatment wetlands, and thus the overall cost, were shown to be most sensitive to predicted ammonia concentrations. A more refined prediction of lower concentrations could eliminate the need for a surface flow wetland and decrease the overall cost.

Testwork has been described that could better define the above conceptual design for the treatment wetlands. These tests should be conducted during the operation of the mine, or shortly after closure, since they will take approximately two years.

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