

## APPENDIX 4A: CONCEPTUAL CLOSURE PLAN

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**CASINO MINING CORPORATION  
CASINO PROJECT**

**CONCEPTUAL CLOSURE AND RECLAMATION PLAN**

December 17, 2013

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## APPENDICES

Appendix A Wetland Water Treatment for the Casino Project

## 1 – INTRODUCTION

### 1.1 OVERVIEW

This document provides a conceptual closure and reclamation plan for the proposed Casino Project to support the environmental assessment process. This plan will be expanded during the Quartz Mining License application to meet additional reporting requirements, such as costing, as detailed in the Yukon Mine Reclamation and Closure Policy.

This closure and reclamation plan is conceptual in nature as Casino Mining Corporation (CMC) is in the process of bringing the mine into operation. Although “conceptual”, this plan is based on numerous site characterization and engineering studies in support of the mine plan and closure. The reclamation plan will be updated periodically throughout the operating mine life (1 year after start of milling and every 5 years thereafter). A final, detailed plan will be submitted at least 2 years before mine closure.

The key documents upon which this plan was prepared are listed as follows:

- Casino Project – Feasibility Study Volume II (M3, 2012)
- YESAB Proposal – Project Description (KP, 2013)
- Water Management Plan (KP, 2013)
- YESAB Water Balance Model (KP, 2013)
- Water Quality Model Report (Source, 2013), and
- Wetland Water Treatment for the Casino Project (Sobolewski, 2013).

### 1.2 PROPERTY DESCRIPTION

The Casino Project is a venture by CMC to develop an Open Pit copper-gold-molybdenum mine in Yukon Territory. The project is located at 62.74°N and 138.82°W, approximately 300 km northwest of Whitehorse, Yukon. A location map is provided on Figure 1.2-1.

The project is located on Crown land administered by the Yukon Government and is within the Selkirk First Nation traditional territory; the Tr'ondek Hwechin First Nation traditional territory lies to the north and Little Salmon Carmacks First Nation to the south.





#### LEGEND

- ★ PROJECT LOCATION
- CITY / TOWN
- EXISTING ROAD
- PROPOSED ACCESS ROAD
- YUKON TERRITORY

#### NOTES:

1. BASE MAP: (C) MICROSOFT BING MAPS AND NATIONAL ROAD MAPS.
2. COORDINATE GRID IS IN METRES.  
COORDINATE SYSTEM: NAD 1983 UTM ZONE 7N.
3. THIS FIGURE IS PRODUCED AT A NOMINAL SCALE OF 1:5,000,000 FOR 8.5x11 (LETTER) PAPER. ACTUAL SCALE MAY DIFFER ACCORDING TO CHANGES IN PRINTER SETTINGS OR PRINTED PAPER SIZE.

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CASINO MINING CORPORATION

CASINO PROJECT

PROJECT LOCATION MAP

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FIGURE 1.2-1

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#### 1.2.1 Physiography

The following descriptions of physiography and climate have been summarized from M3 Engineering and Technology, "NI 43-101 Technical Report - Feasibility Study".

The project is located in the Dawson Range Mountains of the Klondike Plateau. The characteristic terrain features are smooth, rolling topography, with moderate to deeply incised valleys. The deposit area is situated on a small divide. The northern part of the property drains to Canadian Creek and Britannia Creek into the Yukon River. The southern part of the property flows southward via Casino Creek to Dip Creek to the Klotassin River and northward to the Yukon River.

Bedrock outcrop is rare on the property. Soil development is variable ranging from coarse talus and immature soil horizons at higher elevations to a more mature soil profile and thick organic accumulations on the valley floors. The valley bottoms throughout the project area, with impeded drainage, have prominent permafrost features such as peat plateaus, palsas, hummocky tussock fields and polygons.

#### 1.2.2 Climate

The climate in the Dawson Range is subarctic. Permafrost is widespread on north-facing slopes, and discontinuous on south-facing slopes. The climate at the Casino Project is characterized by long, cold, dry winters and short, warm, wet summers, with conditions varying according to altitude and aspect. Streamflow in the region is typically highest during the summer months of May, June, July and August due to a combination of melting winter snowpack, summer rainfall, and melting of the active layer above permafrost. Flows decrease throughout the fall and winter, with minimum flows typically occurring in March or early April.

The mean annual temperature for the Casino Project area is estimated to be -3°C, with minimum and maximum monthly temperatures of -18°C and 11°C occurring in January and July, respectively. The mean annual precipitation (MAP) for the Casino Project area is estimated to be 460 mm (KPLg, 2013).

#### 1.2.3 Seismicity

A probabilistic seismic hazard assessment has been carried out by Knight Piésold Ltd. (KPL) (KPLc, 2012) for the Casino project site. The seismic hazard for the project site is predominantly from shallow crustal earthquakes (magnitudes up to M7) in the region of the southern Yukon, but it is also influenced by the potential for larger magnitude earthquakes (approximately M7.5 to 8.0) occurring farther from the site on the East Denali fault zone.

### 1.3 MINE COMPONENTS

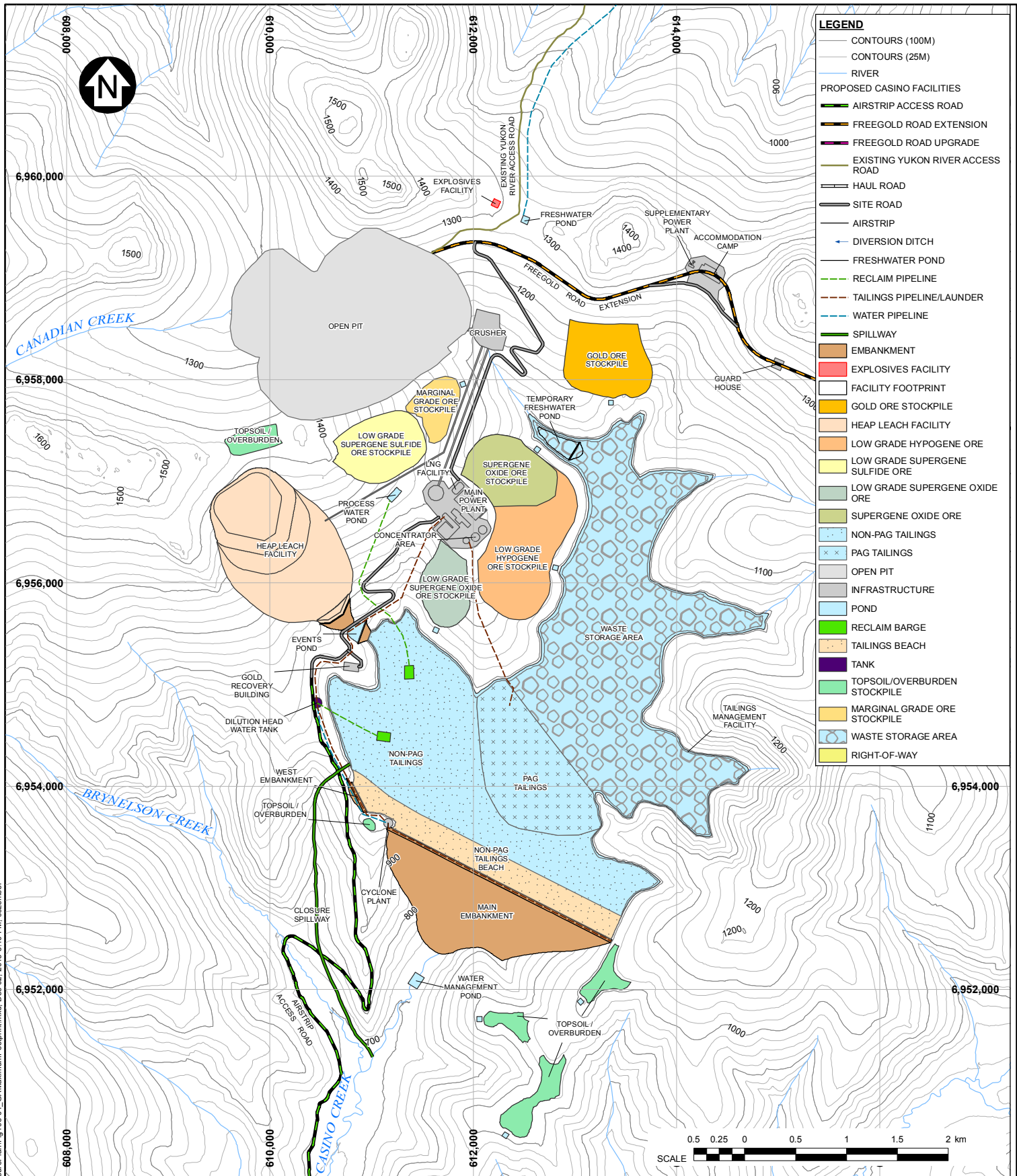
The deposit will be mined using Open Pit methods with a nominal mill throughput of approximately 120,000 tonnes/day of ore over a 22 year operating life. The project components have been categorized as follows:

- Open Pit
- Ore Stockpiles
- Processing Facility, including power and related infrastructure
- Heap Leach Facility (HLF)

- Tailings Management Facility (TMF), and
- Mine Access Road.

Figure 1.3-1 shows the general layout of these mine components at their maximum extent. Section 3 of this plan is divided into the above components. For each component there is a description of the expected final conditions and landscape at closure. The closure objectives are presented, followed by the reclamation activities required to achieve the closure objectives. Following this section, details are provided for closure and reclamation activities, vegetation to be used during reclamation, and purpose and design of the treatment wetlands. The closure plan ends with an overview of closure monitoring and a list of the main indicators to be monitored.





#### NOTES:

1. BASE MAP: RIVERS FROM CANVEC, CONTOURS FROM EAGLE MAPPING.
2. COORDINATE GRID IS IN METRES.  
COORDINATE SYSTEM: NAD 1983 UTM ZONE 7N.
3. THIS FIGURE IS PRODUCED AT A NOMINAL SCALE OF 1:50,000 FOR 8.5x11 (LETTER) PAPER. ACTUAL SCALE MAY DIFFER ACCORDING TO CHANGES IN PRINTER SETTINGS OR PRINTED PAPER SIZE.

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CASINO PROJECT

**GENERAL ARRANGEMENT  
MAXIMUM FOOTPRINT**

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**FIGURE 1.3-1**

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#### 1.4 CLOSURE OBJECTIVES - GENERAL

CMC is committed to achieving the following four objectives during construction, operation, reclamation and closure of the mine:

- Protect public health and safety
- Minimize, mitigate or prevent adverse environmental impacts
- Reclaim the site to a land use state consistent with surrounding conditions, and
- Ensure long-term stability of the spent ore and waste rock storage area and site water quality.

The primary objective of the mine closure and reclamation initiatives will be to achieve physical and geochemical stability of the reclaimed mine components. In addition, the mined landscape should be returned to pre-mining land use and conditions, to the extent practical.

The Yukon Government states “reliance on long term active treatment is not considered acceptable for reclamation and closure planning” (Government of Yukon, 2006). As such, the following specific objectives are considered in all reclamation and closure planning:

- Restoration of the mine area, considering terrestrial restoration (vegetation) compatible with surrounding area
- Physical stability of residual structures (i.e. tailings dam, heap leach facility, etc.)
- Protection of downstream receiving environment, and
- Minimize requirements for post-closure activity (i.e. site presence).

CMC recognizes that a “walk-away” condition is not achievable, but also that reliance on long term “active care”, such as treatment is not acceptable. CMC has designed the Casino Project for “passive care” requiring only minimal management and maintenance during the post-closure period. For clarity, these terms are defined as follows:

**Active Care:** Continuous and ongoing (for decades or centuries) site presence and activity, such as pumping and chemically treating contaminated drainage.

**Passive Care:** Regular or infrequent site presence for monitoring (environmental or geotechnical) and maintenance as necessary (such as repairs to spillways, covers, wetlands). There is no requirement for year-round site presence, or power or chemical reagents for water treatment.

**Walk Away:** This situation occurs when reclamation of a mine, or a component of a mine, is reclaimed such that there is no need for any future activity. For large modern mines this is typically achievable for mine components only, such as buildings, roads or non-PAG waste dumps.

## **2 – APPROACH TO MINE DEVELOPMENT AND CLOSURE PLANNING**

CMC has taken a comprehensive and conservative approach to closure planning in order to ensure that the key objectives of environmental protection are met and that there is no expectation of long-term active care. This has involved a number of key steps as follows:

1. Development of detailed site characterization data base. This has included geology, geochemistry, and receiving environment conditions (water quality and aquatic life).
2. Development of a mine plan which is economically feasible and technically achievable.
3. Evaluation of the potential impacts of the mine plan, followed by modification of the mine plan to minimize potential impacts during and following operations, and to ensure that post-closure active care is not required. This process of modification of the mine plan has been conducted with the ongoing input of mine designers, geologists/geochemists, water quality specialists and mine closure experts. As a result of this process, the mine plan includes the following key elements:
  - a. Design and construction of the TMF dam for the 1 in 10,000 year earthquake and probable maximum flood (the highest standards recommended by the Canadian Dam Association).
  - b. All mine components are conservatively designed to ensure protection of the receiving environment. This includes using first flush or peak concentrations from flooded waste rock, tailings porewater and tailings beaches and embankments for geochemical testing in water quality predictions. Steady – state ML/ARD release rates have been used for ore stockpiles, unflooded waste rock, and pit walls.
  - c. Segregation of the tailings in the mill into non-PAG tailings which is deposited as the tailings beach or used as the source of cyclone sand material for dam construction, and a PAG tailings which is deposited underwater in the pond area of the TMF.
  - d. 98.5% of all PAG waste rock is placed immediately in the TMF for permanent sub-aqueous disposal.
  - e. All PAG rock is placed strategically in the upper portion of the TMF where:
    - i. Seepage pathways are restricted due to the low permeability tailings placed closest to the dam, and
    - ii. The sequence of rock placement allows geochemical reactions which significantly reduce the concentration of metals in seepage and TMF pond water.
  - f. The worst 1.5% of PAG rock (approximately 9 M tonnes) is stockpiled for milling or disposal in the pit at the end of pit operations (i.e. the Marginal Grade Ore Stockpile), which ensures that the TMF pond and seepage water quality is acceptable for long-term passive management.
  - g. All LGO stockpiles are situated within the catchments of the Open Pit and TMF, and resultantly contact water will report to these two locations. However, modelling results suggest that the approximately 30% of groundwater seepage from the Supergene Oxide Low Grade Ore Stockpile could enter the deeper groundwater system and not discharge to the TMF pond, but rather through and beneath the TMF embankment. Therefore, a groundwater collection or infiltration suppression system will be installed to intercept potentially contaminated groundwater issuing from the Supergene Oxide Low Grade Ore Stockpile before it can leave the TMF system.
  - h. Processing of all Low Grade Ore stockpiles is conducted toward the end of mine life which:
    - i. Provides a reclamation surface in the TMF, and
    - ii. Ensures that no PAG material is left on surface at closure.

- i. Design of passive care closure systems (North and South TMF Wetlands, Winter Seepage Mitigation Pond, and the Open Pit gravity discharge system) for water management to mitigate residual water quality concerns that cannot be addressed through modification of the mine plan
- j. An active closure period of 3 years to remove mine infrastructure from the site and construct covers on the TMF embankment, HLF after detoxification, and stockpile footprints. Construction of a seepage mitigation pond downstream of the TMF embankment and two TMF water treatment wetlands will also occur during this period.
- k. A long-term monitoring program consisting of active and passive phases will begin after the primary reclamation activities have been completed. This will consist of: an active post-closure monitoring period of 5 years to ensure that closure elements, specifically TMF wetlands and the winter seepage mitigation pond, are fully functional prior to TMF discharge; and a passive post-closure monitoring phase would then follow, which will continue as long as passive care is needed to ensure environmental protection. A second active-post closure phase may occur beginning when the Open Pit Lake is nearing discharge, to ensure that the Open Pit lake water quality is suitable for release and treatment by the North TMF Wetland. CMC is committed to ensuring all of the above criteria and plans are adhered to throughout the mine life. Comprehensive monitoring and reporting systems will be established in the Regulatory phase to demonstrate that these are being met.

### **3 – CLOSURE AND RECLAMATION PLAN**

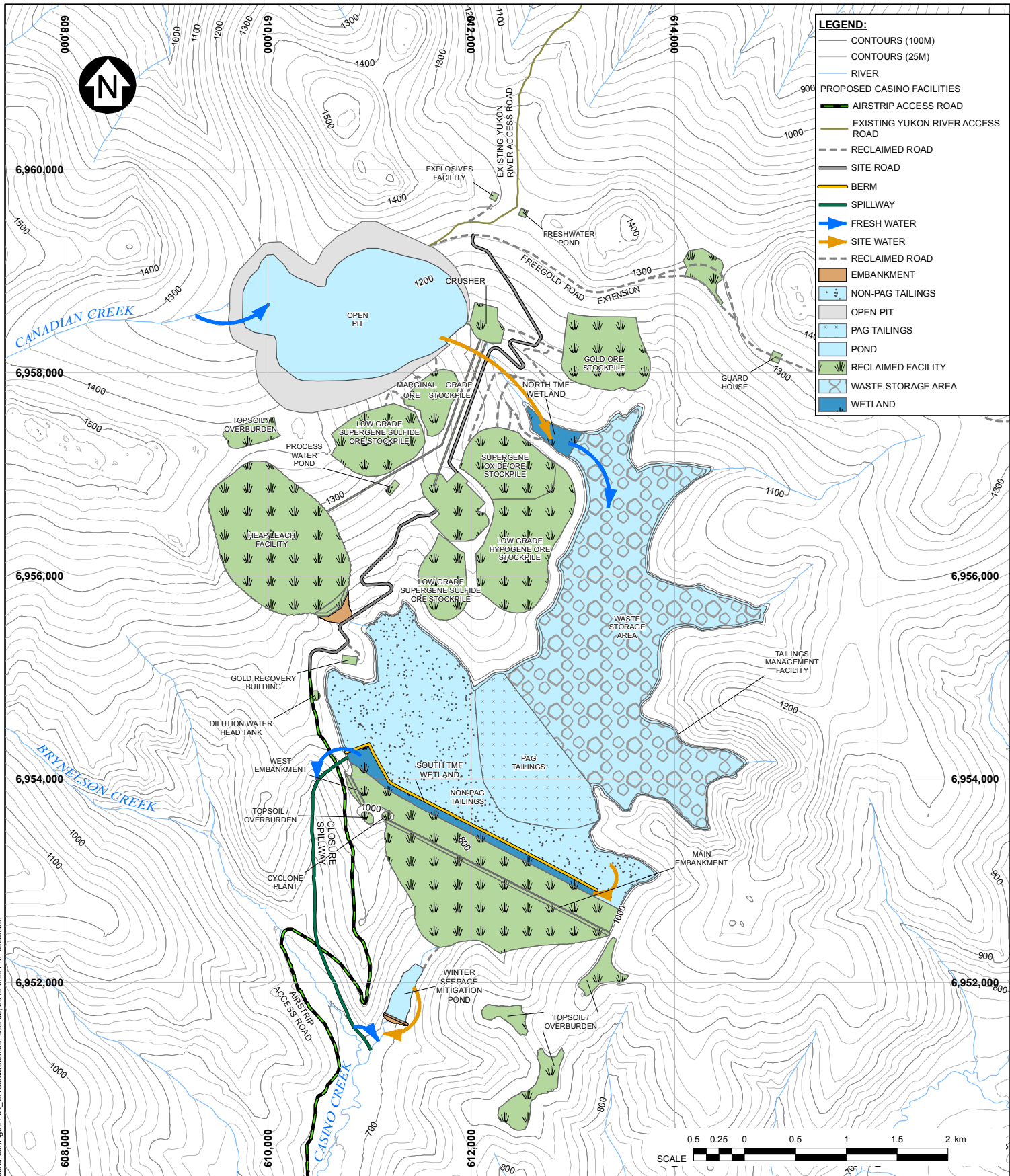
#### **3.1 OPEN PIT**

##### **3.1.1 Overview**

The Open Pit will be located between the headwaters of Casino Creek and Canadian Creek and will occupy an area of approximately 300 ha. Approximately 1.78 billion tonnes of material will be removed in five stages over 22 years (i.e. 3 years of pre-production and 19 years of operations). The Open Pit is shown on Figure 3.1-1.



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FIGURE 3.1-1			REV 0	

### 3.1.2 Final Landscape at Closure

The proposed Open Pit will extend to a maximum depth and width of approximately 600 m and 2400 m respectively resulting in an excavated surface area of 3.14 km<sup>2</sup>.

During operations, a diversion ditch will be installed along Canadian Creek (starting in about Year 10) to direct the main Canadian Creek channel around the pit.

### 3.1.3 Reclamation Activities

Open Pit mining will cease in Year 19 and LGO will be processed through the mill until the end of Year 22. Open Pit dewatering will be discontinued once active mining stops in Year 19. The closure phase for the Open Pit will therefore begin in Year 19.

The closure plan is to flood the pit. The Canadian Creek diversion channel will be breached to direct flow into the pit, as it is desired to take advantage of the high alkalinity in the creek water, as well as to accelerate filling to minimize potential oxidation of the pit walls and preclude the need for maintenance of permanent diversions. It is estimated that it will take approximately 95 years to flood the pit to the point of discharge to the TMF. Accelerated filling of the pit by pumping from the Yukon River was evaluated as a potential closure strategy. Analyses showed that this would not materially affect the pit water quality at the time of overflow and is therefore not planned for closure.

Upon filling, the pit walls will be up to 200 m above the surface of the lake and the exposed surface area will be approximately 1.98 km<sup>2</sup>. Rocks consisting of oxide cap (NAG), supergene (PAG), and hypogene (PAG) will be exposed in the pit wall at the end of operations. Most of the hypogene will be submerged.

When the pit lake begins to overflow approximately 95 years after mining has ceased, the water quality is predicted to be pH neutral, and hence will not require treatment for acidic drainage. However, pit lake water is predicted to have metal concentrations which could cause impacts to the receiving environment and therefore overflow will be directed to the North TMF Wetland for treatment. Details of the exceeding parameters and wetland design and treatment capabilities are provided in Section 6 and Appendix A.

Overflow from the pit will flow by gravity via a decant pipe buried in a sealed off trench as shown in Figure 3.1-2. A valve on the pipe will control the rate and timing of pit lake discharge. The pit lake overflow elevation is 1095 m. The pipe invert elevation will be designed to allow for storage of a 100 year return period freshet plus 5 m of freeboard to prevent leakage through bedrock. The pit lake discharges (totaling approximately 2 Mm<sup>3</sup> per year) will be released to the TMF wetland at a controlled rate during the warmest months of the year (June through September). The average rate of discharge is calculated to be approximately 180 l/s (KP 2013, YESAB Water Balance Report).

The pit lake decant system will not require any pumps or power system to operate. Twice a year the decant valve will be opened or closed. It is envisioned that this could be done remotely using solar power to operate the valve and monitor flows.

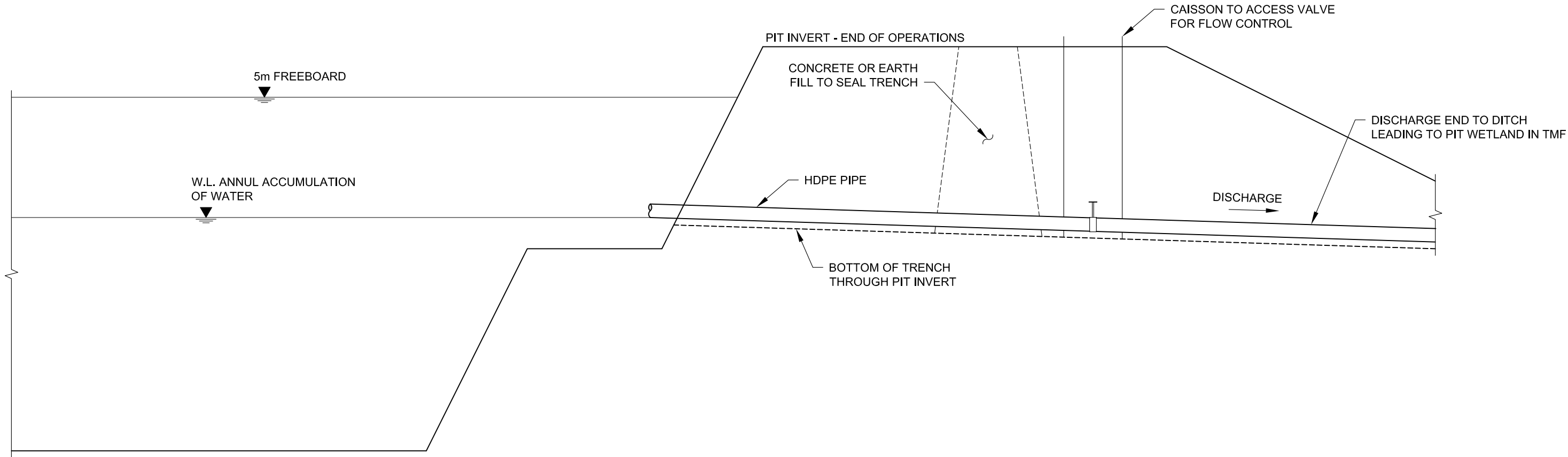
An allowance has been provided for placing any remaining unprocessed LGO material into the Open Pit. This is in addition to the planned 8.6M tonnes of acidic Supergene rock. The volume of this material was assumed to be 7.2 Mt, or 5% of the total LGO volume accumulated during the mine life. This material will be assayed before placement in the pit, and if it is predicted to adversely affect the

pit water quality such that it could not be treated in the TMF wetland, then it will be amended with lime during placement in the pit.

As is typical for Open Pits, the pit walls will be steep and unsafe to access. A berm will be constructed at potential access points around the perimeter of the pit to prevent human and wildlife access. Road access will be decommissioned to limit human access.

**Table 3.1–2 Summary of Open Pit Reclamation Activities**


- |  |
|--|
| <ul style="list-style-type: none"> <li>• Remove diversion ditches.</li> <li>• Remove any remaining infrastructure or equipment within the Open Pit.</li> <li>• Construct berm at access points around perimeter of Open Pit.</li> <li>• Clean up and dispose of any debris and garbage.</li> <li>• Construct decant system that leads to North TMF Wetland.</li> <li>• Conduct post-reclamation water quality monitoring.</li> </ul> |
|--|



**PIT GRAVITY  
DISCHARGE SYSTEM**  
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CASINO PROJECT		
OPEN PIT GRAVITY DISCHARGE SYSTEM		
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	<b>FIGURE 3.1-2</b>	REV 0

### 3.1.4 Closure Monitoring

Approximately 1.1 Mm<sup>2</sup> of PAG and oxide rock is expected to be exposed on pit walls above the ultimate flood elevation. As a result, overflow from the pit lake is predicted to have neutral pH and slightly elevated concentrations of metals, including SO<sub>4</sub>, Cd, Cu, Fe, Mo, Se, and U (Source, 2013). Any trace amounts of cyanide (WAD-CN) within the HLF draindown water, which is pumped to the Open Pit during HLF decommissioning, will naturally decay to non-detectable levels before pit overflow.

A water quality monitoring program will be initiated at the start of pit flooding. The purpose of the program will be to validate the predictions of water quality and identify if any changes are required in order to achieve the passive care closure scenario. Potential changes, depending upon the nature of any deviations from predicted water quality are:

- Change in the size of the North TMF wetland
- Batch (chemical) treatment of the pit water to reduce concentrations of select COC's (such as copper or selenium)
- In-situ bio-treatment to remove select COC's
- Stratification of the pit lake to improve surface water quality prior to discharge, and
- If better water quality develops deeper in the pit lake, the decant system can be modified to allow removal of the deeper water (instead of decanting surface water).

## 3.2 ORE STOCKPILES

### 3.2.1 Overview

The types of temporary stockpiles, their description and total tonnage are provided in Table 3.2–1.

**Table 3.2–1 Temporary Stockpile Description**

Temporary Stockpile	Description	Total over Mine Life (tonnes)
Gold ore stockpile	Crushed gold ore intended for the heap leach stored Year -3 to Year 15	56,674,000
Low-grade ore stockpiles <ul style="list-style-type: none"> <li>• supergene oxide ore</li> <li>• hypogene ore</li> <li>• supergene sulfide ore</li> </ul>	Low grade ore is stockpiled up to Year 17 and milled during the last 4 years of operation, Year 19 to 22	14,043,000 90,433,000 39,351,000
Supergene oxide (SOX) ore stockpile	The SOX ore is stockpiled during the construction phase and in Year 1 of operations for processing during Years 4-12 together with direct feed mill ore.	32,410,000
Marginal grade ore stockpile	Acidic supergene waste rock that cannot be disposed of in TMF without adversely affecting TMF water quality will be milled or disposed of in the pit.	8,837,000

**NOTES:**

1. Adapted from the Casino YESAB Proposal – Project Description



The closure objectives for the ore stockpile areas are:

- Remove all stockpiled ore from each area
- Ensure material within the footprint or downslope is not a source of contaminated runoff/drainage, and
- Restore the footprint to the original topography, replace topsoil and vegetate.

In addition to the ore stockpiles, there will be several stockpiles of topsoil and overburden located around the mine site that will have been salvaged during general site preparation for use in reclamation activities.

### 3.2.2 Final Landscape at Closure

A general arrangement of the project site showing the location and maximum extent of the stockpile footprints is provided in Figure 1.3-1. At closure all material will have been processed from the stockpiles and the surfaces restored and vegetated.

### 3.2.3 Reclamation Activities

The closure plan assumes that some LGO material will not be suitable for processing. This material will be placed in the Open Pit below the flood elevation. An allowance is made for 5% of the total volume of LGO rock (7.2 Mt) to be relocated into the pit after closure of the mill. This provision also includes removal of soil/rock in the footprint of those piles, which may have become contaminated due to seepage from the overlying oxidizing rock.

Following removal of low grade ore and contaminated material from the footprint of the stockpiles, the surface will be covered with topsoil salvaged during mine development, and revegetated.

During operations, seepage from the low grade supergene oxide ore will be intercepted with groundwater collection wells or a groundwater infiltration suppression system. All other seepage and surface runoff will flow to the TMF. After the low grade supergene oxide ore is processed (Year 22) the groundwater seepage mitigation system will be operated until groundwater seepage water quality is acceptable and then it will be decommissioned. After reclamation, all runoff from the former stockpile areas will drain by gravity into the TMF.

**Table 3.2–2 Ore Stock Pile Reclamation Activities**

<ul style="list-style-type: none"> <li>• Remove and dispose of residual LGO</li> <li>• Remove and dispose of contaminated material from footprint of stockpile areas</li> <li>• Temporary operation of the low grade supergene seepage system</li> <li>• Cover disturbed areas with topsoil and re-vegetate</li> <li>• Monitor runoff from former stockpile areas</li> <li>• Decommission seepage pump systems</li> <li>• Conduct geochemical analysis of all material which is to be placed in the Open Pit. Update pit water quality model accordingly.</li> <li>• Monitor vegetation recovery.</li> </ul>
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### 3.3 PROCESSING FACILITY & INFRASTRUCTURE

#### 3.3.1 Overview

The major infrastructure that comprise the plant site, and that will require decommissioning during closure include:

- Milling infrastructure: two primary crushers, one for oxide ore and one for sulphide ore; 1.6 km conveyor belt; concentrator circuits including grinding mills, flotation circuits, reagent mixing, storage and distribution systems, concentrate filter facility and thickeners.
- Truck shop (~2.4 ha).
- 1,000-person residence camp (1,000 people during construction 600-700 during operations).
- Operation support facilities: administration building, mine dry, laboratory, warehouse, laydown area and light vehicle maintenance building (pre-engineered steel structures).
- Guard shed/scale house (modular construction).
- Access roads not required for inspection and monitoring.
- Power generation infrastructure, LNG receiving, storage and distribution facilities and power distribution infrastructure.

#### 3.3.2 Final Landscape at Closure

A general arrangement of the project site showing the location of the processing facility and related infrastructure is provided in Figure 1.3-1. At closure all material will have been processed and the facilities will be ready for decommissioning.

#### 3.3.3 Reclamation Activities

The following reclamation activities are planned for closure:

- Remove (or cut to surface) all infrastructure (i.e. buildings, concrete walls, pipelines, tanks, bridges, culverts, electrical). It is assumed that facilities and equipment will have little salvage value at the end of the mine life. All facilities will be cleaned of hazardous materials, leaving only inert materials (metal, plastic, glass, concrete) for demolition. Recycling will be conducted to the extent which is viable and practical.
- Hazardous materials will be shipped for off-site disposal.
- Inert demolition material will be disposed of in an industrial landfill at the uphill side of the heap leach facility. Once landfilling is completed, leached rock on the heap will be dozed over the

material to provide burial. A 0.5 m cover of topsoil will be placed on top of the crushed rock cover over the demolition waste.

- Any potentially contaminated soil resulting from fuel and lubricant storage will be excavated, bio-remediated on site and then placed in the industrial landfill for disposal.
- Assorted mining equipment (i.e. trucks, shovels, fuel drums) will be disposed of in the industrial landfill following removal of hazardous materials (liquids, batteries).
- Decommission the main power plant and switch to backup power appropriately sized for closure activities. Decommission backup power when reclamation activities are complete.
- Concrete foundations will be fractured to allow drainage and covered with topsoil and revegetated.
- All disturbed areas will be rehabilitated with topsoil as necessary and re-vegetated.
- Roads not required for post-closure activities will be reclaimed by removing culverts and safety berms, re-establishing natural drainage channels, scarifying the surface and re-vegetating.

It is estimated that approximately 1.7 Mm<sup>3</sup> of topsoil will be required to cover disturbed areas of the processing facility and infrastructure. Further details regarding topsoil covers are provided in Section 3.7

**Table 3.3–1 Processing Facility and Infrastructure Reclamation Activities**

<ul style="list-style-type: none"> <li>• Dismantle buildings and infrastructure and salvage any material with value. Inert material without salvage value disposed of in on-site landfill.</li> <li>• Remove and properly dispose of any hazardous materials off-site</li> <li>• Decommission power plant.</li> <li>• Reclaim roads not required for post-closure activities.</li> <li>• Remove and remediate any hydrocarbon contaminated soils.</li> <li>• Re-vegetate disturbed areas.</li> </ul>
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### 3.4 HEAP LEACH FACILITY

#### 3.4.1 Overview

The Heap Leach Facility (HLF) will process approximately 157.5 Mt of crushed gold ore with a cyanide leaching system, starting in pre-production (3 years prior to mill start-up) and continuing to Year 18. The HLF is located in a small valley about 1 km south of the pit and within the catchment area of the TMF. An earthen confining embankment at the eastern end of the pad will provide structural support and create the in-heap solution storage volume needed for operations.

The HLF foundation will be prepared by excavation of all soil to stable bedrock. It will have a composite liner system comprising an LLDPE liner, compacted soil liner and leak detection and recovery system. A double composite liner system will be installed in the lower portion of the leach pad for areas with potential for solution storage. The HLF will be constructed in five stages, with the final pad having a surface area of approximately 1.5 Mm<sup>2</sup>.

In heap storage capacity will total 172,600 m<sup>3</sup> and an events pond with a capacity of 74,400 m<sup>3</sup> designed for temporary storage of storm runoff and pregnant solution overflow during shut down will be constructed at the foot of the HLF confining embankment.

During operations, diversion ditches route surface runoff around the HLF to the events pond to provide makeup water for the leachate solution.

Gold ore to feed the HLF will be located in a temporary stockpile east of the Open Pit, close to the crusher (see Section 2.2).

#### 3.4.2 Final Landscape at Closure

The final heap leach pad required for an ore tonnage of 157.5 million tonnes will have a surface area of approximately 1.5 Mm<sup>2</sup> and a total height of 150 m. This includes lift benches 8 m in height at repose face angles of approximately 1.4H:1V, with an approximate 9 m setback from the edge of the slope below resulting in an overall downslope angle of 2.5H:1V.

#### 3.4.3 Reclamation Activities

Upon cessation of supplemental gold recovery at the end of Year 18, the heap will be detoxified through cyanide removal by rinsing with treated solution and/or freshwater for 5 years (Years 19 to 23) using the solution irrigation system.

Following the end of rinsing, the water accumulated in the heap will be allowed to draindown until all the ore on the heap reaches the long-term estimated moisture content of 5% (by mass). This water will be pumped to the Open Pit.

Modeling indicates that draindown water may have sufficiently elevated concentrations of selenium and mercury that the ultimate discharge of pit water to the North TMF Wetland could result in bioaccumulation of these parameters and cause impacts to waterfowl. Different options for addressing elevated levels of mercury and selenium are available. For example, accounting for the precipitation of selenium with sodium sulfide during the gold recovery SART process would reduce selenium levels in the re-circulated HLF water, and therefore also the draindown water. It may also be possible to treat mercury and selenium during operations by adding additional processes to the gold recovery circuit; however, the passive treatment option currently selected for closure design of the Casino Project is a bioreactor. Mercury and selenium would both be removed from HLF draindown prior to it entering the open pit, which would protect the North TMF Wetland from being exposed to their elevated concentrations. Selection of this mitigation treatment option is decidedly conservative from a closure design and costing perspective, and was selected for this reason along with the effectiveness and passive nature of the treatment. The bio-reactor details and relevant case history are described further in Appendix A. Following draindown, the bioreactor will be shut down and the matrix will be permanently sealed in place, such that future release of the selenium precipitate cannot occur.

Once the draindown quality and flow reach acceptable levels (Year 29), the heap surface will be reclaimed with a closure cover to reduce infiltration, all pumping systems and upstream diversion ditches will be decommissioned, and runoff from the HLF will discharge naturally to the TMF pond.

The design of the closure cover will effectively reduce infiltration into the heap to 80% of mean annual net precipitation. This will be achieved by re-contouring the surface of the heap to achieve surface slopes between 10:1 and 5:1, and then placing a 0.75 m cover of low permeability material on top of the heap. The cover is required to reduce the selenium load discharging from the closed

facility to the TMF. If water quality monitoring during operations and closure of the HLF shows that Se loads are less than predicted, the design of the cover will be re-evaluated.

#### 3.4.3.1 Water Treatment

The water treatment technology proposed to reduce cyanide concentrations of toxic weak acid dissociable cyanide (WAD Cn) to levels acceptable for discharge is the Inco-SO<sub>2</sub> process. The efficacy of this process was established in a laboratory evaluation of the treatability of solutions produced by test work conducted to evaluate the heap leach process (R&C Environmental Services, 2010). Based on the results of that study, inputs from the water balance model and professional experience (per. comm. J. Marsden, 2013), the following assumptions have been utilized in the preparation of the water quality model for the Project:

26 m <sup>3</sup> /hr	Cyanide destruction plant capacity during operations, based on the size required to treat 75% of the event pond capacity within 3 months, assuming primary and secondary treatment will be required to meet the discharge criteria to the TMF pond. Any solution suitable for reuse would be recycled in the heap process (assume 25%).
1,312 m <sup>3</sup> /hr	Cyanide destruction plant capacity during rinsing - set to match the operational irrigation rate.
2 ppm WAD Cn	Cyanide destruction plant criteria for discharge to the TMF pond during operations.
5 ppm WAD Cn	Cyanide destruction plant criteria for discharge to the Open Pit during closure (draindown). This value was used in the Water Quality Model.
0.03 ppm WAD Cn	HLF discharge quality during the 10-year period following draindown (Year 29-38).
0.0 ppm WAD Cn	HLF discharge quality during post-closure (Year 39-perpetuity).

Note that there likely would be a gradational transition between rinsing and draindown, such that during the later stages of rinsing, solution which is suitable for discharge to the pit would be discharged rather than recycled. Towards the latter stages of draindown, no recycling takes place.

The post-closure water quality discharges calculated for the HLF have elevated concentrations of certain metals, including SO<sub>4</sub>, Cd, Cu, Fe, Mo, Se, and U (Source, 2013). This water will discharge to the TMF Pond, where it will be diluted and subsequently treated by the South TMF Wetland prior to discharge to the downstream receiving environment. There will be no exceedances of water quality criteria for Cn in Casino Creek.

**Table 3.4-1 Heap Leach Facility Reclamation Activities**

- Detoxification followed by draindown of HLF.
- Grading, covering and re-vegetation of final heap slopes. The cover will be designed to reduce infiltration to 20% of mean annual net precipitation, and establish vegetation.
- All upstream diversion ditches will be decommissioned and any excess runoff from the HLF will discharge naturally to the TMF.



#### 3.4.4 Closure Monitoring

A water quality monitoring and seepage detection program will be in place during operations of the HLF, and the water quality monitoring program will be continued into closure. The purpose of the program will be to determine when CN concentrations have reached acceptable levels to discontinue use of the CN treatment plan, and to validate the water quality predictions. If water quality is different than predicted, potential changes that may be required in order to achieve the passive care closure scenario include:

- Treatment of draindown water in a bioreactor may no longer be required prior to it being pumped to the Open Pit.
- Treatment of long-term runoff in a bioreactor before it enters the TMF.
- Winter storage of heap seepage (no release) in pregnant solution zone of heap, with discharge and treatment via bioreactor during biologically active months.
- Event ponds to remain for contingency storage.

### 3.5 TAILINGS MANAGEMENT FACILITY

#### 3.5.1 Overview

All PAG waste rock, tailings and supernatant water from the mining process will be stored in the Tailings Management Facility (TMF) located southeast of the pit within the upper Casino Creek valley (Figure 3.1-1). The TMF will cover approximately 1,120 ha. It is designed to retain 956 Mt of tailings and up to 658 Mt of PAG waste rock and overburden materials.

#### 3.5.2 Waste Storage Area (WSA)

Throughout operations, waste rock will be placed at an elevation above the tailings and pond water level to provide a dry, stable placement surface. At the end of operations (Year 19 to 22), LGO will be processed with de-pyritized LGO tailings discharged over the waste rock, to an average depth of 3 m. The WSA will then be leveled to ensure that all tailings and waste rock are below the closure invert elevation of the spillway, and will therefore remain permanently submerged.

#### 3.5.3 Tailings

Approximately 80% of the total tailings will be non-PAG material, following removal of the pyrite component. These tailings will be used for the production of cyclone sand for construction of the Main Embankment. In addition to cyclone sand, the cyclone plant will produce a high volume, low solids content stream, containing the fines fraction of the tailings. This material will be discharged into the TMF to form the beach along the crest of the dam.

Approximately 20% of the total tailings will be PAG tailings which are to be delivered to the TMF in a separate pipeline. The PAG tailings will be deposited below water in the central region of the TMF by discharging the tailings from the north-western side of the TMF, close to the WSA. Deposition in this area will keep the PAG tailings within the submerged portion of the tailings beach. This will minimize seepage from the PAG tailings and ensure that they remain in a subaqueous state.

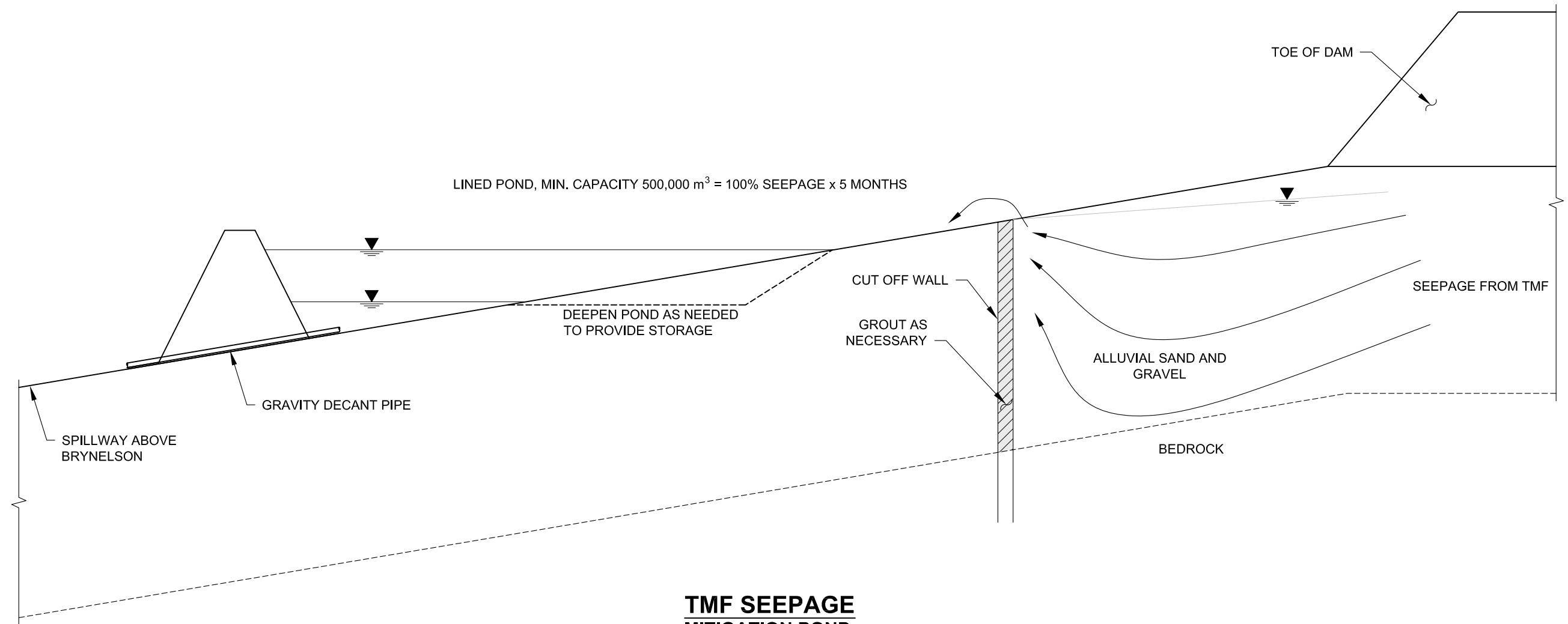
#### 3.5.4 Water Management Pond

During operations, a water management pond will be located downstream of the Main Embankment to recover seepage from the dam. This water is needed for operations and is a less costly source than the Yukon River. A booster pump and pipeline will transfer water from this pond back to the TMF pond. In closure, the TMF water management pond will be replaced by a larger pond designed to collect and store seepage from the TMF during the winter months. This will prevent water quality issues in Dip Creek during the low flow winter months.

The winter seepage mitigation system is shown in Figure 3.5-1. It consists of 3 components:

- An upstream cut-off wall keyed into bedrock to intercept all seepage from the dam and force it to surface where it will drain by gravity into the storage pond.
- A storage pond lined with LLDPE or HDPE membrane with a capacity of approximately 600,000 m<sup>3</sup>. This pond will have capacity for 100% of the winter seepage flow, plus additional capacity for minor inflows from the watershed between the cutoff wall and TMF embankment, plus freeboard. (35 l/s x 5 months = 450,000 m<sup>3</sup>).
- An earth dam approximately 10 m high, located downstream of the storage pond and upstream of the TMF closure spillway. This dam will incorporate a gravity decant pipe system which will be closed for the winter months and opened in the spring freshet.


The winter seepage mitigation pond will not require any pumps or power system to operate. Twice a year the decant valve will be opened or closed. It is envisioned that this could be done remotely using solar power to operate the valve and monitor flows. The pond will collect seepage from December through April and then discharge at a rate of approximately 130 l/s beginning in May after the onset of the spring freshet.



**TMF SEEPAGE  
MITIGATION POND**  
NTS

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0	17DEC'13	ISSUED WITH REPORT	KT	PP	DG	KJB
REV	DATE	DESCRIPTION	DESIGNED	DRAWN	CHK'D	APP'D

CASINO MINING CORPORATION		
CASINO PROJECT		
WINTER SEEPAGE MITIGATION POND		
	P/A NO. VA101-325/14	REF NO. 1
	<b>FIGURE 3.5-1</b>	
		REV 0

### 3.5.5 Closure Objectives

Closure objectives for the TMF are to establish acceptable water quality for long-term discharge to the receiving environment without the requirement for active water treatment. Post-closure, the TMF will be required to maintain long-term geotechnical and geochemical stability of the mine waste rock and tailings, and to manage surface water to protect the downstream environment.

The following objectives have been considered in development of the TMF closure plan:

- Meet Canadian Dam Association (CDA) standards for design. A CDA hazard classification of “high” has been determined (KP, 2012 TMF Feasibility Design). For closure Inflow Design Flow designed as Probable Maximum Flood.
- Ensure protection of downstream receiving environment for all water released from site by way of long-term, passive water treatment.
- Minimize long term requirements for maintenance.

It is proposed to achieve these objectives through the following TMF reclamation initiatives:

- Decommission the water management pond and pumping system from the toe of the TMF embankment and replace with a larger seepage mitigation pond and gravity discharge system.
- Construct an overflow channel (spillway) to allow surface water to discharge downstream of the TMF, once desired water quality has been established.
- Construct engineered wetlands within the TMF to provide passive water treatment systems for removal of contaminants from Pit Lake and TMF overflow.
- Remove access roads, ponds, ditches, pipelines, and borrow areas not required Post-Closure.

### 3.5.6 Final Landscape at Closure

The TMF will cover about 11 km<sup>2</sup>, and will be comprised of the TMF dam, beach, a non-PAG tailings area, a PAG tailings area, a waste storage area and the supernatant pond, as shown on Figure 1.3-1. Only non-PAG materials will be exposed within the beach. The crest of the tailings dam will be at an approximate elevation of 998 m with a maximum height of 286 m. The WSA will be completely covered by tailings and the supernatant pond in Year 20 of operations. Tailings will naturally consolidate during operations and for a period of time after closure until excess pore water pressures have dissipated. The majority of consolidation is expected to occur within 5 years following operations.

### 3.5.7 Reclamation Activities

The principal reclamation activities are construction of the TMF closure spillway and winter seepage mitigation pond, development of wetland treatment systems within the TMF for passive treatment of pit lake water and tailings supernatant water, and pumping of the TMF pond to the Open Pit to improve pond water quality prior to post-closure. Potentially, up to 53 Mm<sup>3</sup> of water may be added to the pit. The lowered pond surface during pumping (Years 23-27) will also allow for the construction of the TMF wetlands. Construction and commissioning of the wetlands will be complete prior to discharge of the TMF pond down the closure spillway in Year 31. Field trials will be constructed during operations to establish the most effective plants and substrate systems for the wetlands. Further details regarding wetland design are provided in Section 6 and Appendix A.

TMF pond discharge will be routed through the South TMF Wetland for treatment prior to discharge down the spillway. The spillway will consist of a 20 m wide sill excavated through bedrock and located close to the west abutment of the West Saddle Embankment.

TMF pond elevation is critical such that wetlands are neither excessively submerged nor dry. Wetlands naturally experience water level fluctuations; however some moderating of the levels may be needed to ensure good performance of the wetlands. Maintenance of water level fluctuations within acceptable limits requires capacity for:

- Natural seepage/inflow from TMF catchment
- Spring and storm inflow surge

To control water levels and water quality the invert of the spillway will be located to maintain approximately 0.5 meters of water cover within the wetlands. If required, a concrete weir (stop-log type system) or a buried decant system (similar to the pit decant) could also be installed in the spillway structure to allow regulation of the water level in the TMF pond.

Discharge from the spillway will be routed to the south, ultimately terminating at a plunge pool prior to discharging to Casino Creek just upstream of Brynelson Creek, as shown in Figure 3.1-1. The discharge channel will be aligned roughly parallel to Brynelson Creek. Rip-rap will be placed within the channel to maintain long term stability.

After commissioning of the winter seepage mitigation pond and South TMF Wetland, and discharge of the TMF pond, the sumps and pumps of the water management pond will be decommissioned. The groundwater monitoring wells and all other geotechnical instrumentation will be retained for use as long term monitoring devices.

As well as water management, physical stability of the TMF will consider the long term stabilization of all exposed, erodible materials and will include the following:

- The beach will be reclaimed, covered with topsoil (approximately 0.5 Mm<sup>3</sup>) and vegetated.
- The embankment will be reclaimed, covered with topsoil (approximately 1 Mm<sup>3</sup>) and vegetated. A 0.3 m deep cover of rip-rap will also be considered as an alternative, if it is determined that additional erosion protection is required.

### 3.6 AIRSTRIP AND SITE ACCESS ROADS

The airstrip and road from the airstrip to the mine site will not be decommissioned and will be maintained in order to provide access to the site for inspections and monitoring during closure. Ancillary facilities such as the hangar building and fueling storage will be decommissioned. Any potentially contaminated soil resulting from fuel and lubricant storage will be excavated, bio-remediated on site and placed in the industrial landfill for disposal.

### 3.7 COVER MATERIALS AND QUANTITIES

Overburden and topsoil will be stripped from the Open Pit, heap leach facility and TMF embankment footprints during construction. The material is expected to be made up of colluvium (silty sand with gravel) and topsoil. Some of the excavated material will be used in the construction of the embankments for the heap leach facility and TMF; however, a large quantity of overburden and topsoil will remain and will be stockpiled for use in future reclamation activities.

The estimated soil quantities available for use as closure cover material are summarized below.



**Table 3.7-1 Volume of Cover Material Available (Mm<sup>3</sup>)**

Facility	Topsoil Volume	Mineral Soil Volume	Total Volume
Open Pit	0.9	0	0.9
Heap Leach Facility	0.9	3.3	4.2
Low Grade Ore Stockpiles	0.8	9	9.8
TMF Embankment	1.6	8.4	10.0
Total Volume	4.2	20.7	24.9

As shown on Figure 3.1-1, overburden and topsoil stockpiles are located north of the heap leach facility and south of the TMF embankment. Topsoil will be excavated and stored separate from the overburden when practical.

All covers are designed to be constructed of topsoil at an average thickness of 0.5 m. The topsoil will help reduce soil amendment requirements and provide a natural seed bank, which will help encourage re-vegetation. The estimated soil requirements for closure covers are summarized below.

**Table 3.7-2 Volume of Topsoil Required**

Facility	Surface Area (Mm <sup>2</sup> )	Cover thickness (m)	Volume (Mm <sup>3</sup> )
Ore Stockpiles	4.0	0.5	2.0
Processing Facility and Infrastructure	3.4	0.5	1.7
Heap Leach Facility	1.5	0.5	0.8
TMF Embankment	2.0	0.5	1.0
Subtotal			5.5
10% required for maintenance			0.5
Total Volume			6.0

The volume required exceeds the volume of topsoil available. In order to make up this deficit, overburden will be mixed with mineral soil and amended with nutrients as necessary. The stockpiles have sufficient capacity to store this volume.

### 3.8 FREEGOLD ROAD

Following the completion of mining and the active phase of closure activities, the Freegold Road Extension will be decommissioned. Decommissioning of the road will ensure that future vehicular access to the mine site will not be possible. The public portion of the existing Freegold Road will remain open for public use under the ownership and maintenance of Yukon Highways and Public Works.

The objectives of road decommissioning will be to stabilize the road footprint and restore natural drainage patterns while maintaining water quality and reducing the risk of landslides. The degree of decommissioning activities required to achieve these objectives will vary depending on characteristics of each road segment. Factors such as slope failure risks, safety hazards, erosion potential, water quality, water quantity, and fish habitat proximity will all influence the chosen mitigation strategies. Typically the road decommissioning will include most, if not all of the following activities.

- All bridges, stream culverts, and surface drainage cross-culverts along the road will be carefully removed. Removal of stream culverts and bridges may require restoration of the natural stream channel width and gradient, and armouring of the stream banks with rock. Work in fish-bearing streams will occur during timing windows that minimize fish impacts as prescribed by the Department of Fisheries. Cross-culverts will be removed and replaced with cross-ditches to move surface runoff from the road top and roadside ditches to non-erodible soils downslope. Cross-ditches located on longitudinal grades will require ditch blocks installed to intercept ditch runoff. Cross-ditches located at natural low spots will not require ditch blocks and will be broader with gentler slopes to capture the converging runoff. Rock armouring will be placed at all cross-ditch outlets. Cross-ditches will be prepared for natural re-vegetation and may be planted or seeded with local species to prevent erosion of exposed fine grained soils.
- Along the entire length of the road, the top surface will be scarified and left in a condition that promotes natural re-vegetation. Any available local windrowed topsoil may be re-used on the surface and seeding or planting of local species may be completed along the road where appropriate.
- Where the road is located on steep side slopes or potentially unstable terrain, slope angles may need to be restored by pulling back side cast material on select sections of road to reduce the risk of slope failure. Any retaining walls and potentially unstable fills will be removed. Waterbars, berms or outslowing of the remaining road structure may also be required in some areas to intercept water running down the road and divert it to the stable slopes below. Steep slopes will be revegetated to improve slope stability and re-establish natural vegetation successional pathways.
- Where the road is located in valley bottoms or on stable terrain with gentler side slopes, road fills are expected to be stable and will remain in place. Re-sloping of the road top will be completed in select locations to control surface run-off, limit erosion of fine grained soils, and facilitate the removal of culverts and bridges.

Further details of road decommissioning will be developed as part of the detailed project decommissioning planning and in accordance with the requirements of an approved road management plan.

## **4 – CLOSURE AND RECLAMATION SUMMARY TIMELINE**

### **4.1 OPEN PIT CLOSURE**

The following is a summary of the Open Pit closure activities described previously in Section 3.1 presented as a timeline.

#### **Year 19 to 114**

- Breach Canadian Creek diversion channel to allow flooding of Open Pit (Year 19).
- TMF supernatant pumped to pit while closure activities are carried out in TMF (Years 23 – 27).
- Deposit 8.8 Mt of marginal grade ore and 7.3 Mt of LGO rock and foundation material into the Open Pit (Years 19 - 25). Amend with lime if required to achieve acceptable water quality.

#### **114 + years**

- Pit filled to elevation of overflow
- Install decant system to discharge overflow from the pit lake to the TMF
- Summer decant of water to wetland in TMF

### **4.2 TMF CLOSURE**

The following is a summary of the TMF closure activities described in Section 3.5 presented as a timeline in Project Years.

#### **Year 23 – 27**

- Remove temporary mine infrastructure (mill, camp, pipelines, etc.) (Years 23 - 25)
- Reclaim and cover site roads, HLF, TMF embankment, ore stockpile footprints and mine infrastructure footprints (Years 23 – 25)
- TMF supernatant pumped to pit
- North and South TMF Wetlands constructed
- Winter seepage mitigation pond constructed (Years 23 - 25)
- Continue pump back of TMF seepage to the TMF pond
- Continue environmental program (sampling/analysis/reporting)

#### **Year 28 – 30**

- Wetlands are grown to full productivity in TMF,
- When appropriate; validation of wetland performance with pump back seepage water and/or pit water will be conducted
- Continue pump back of TMF seepage to TMF pond
- Continue environmental program (sampling/analysis/reporting)

#### **Year 31 – 114 (TMF Discharging)**

- TMF discharge treatment using passive biological system (South TMF Wetland).
- TMF Seepage stored in winter seepage mitigation pond during December through April. Pond discharged via gravity decant during high dilution summer runoff period.
- Pit water quality monitored and amended if necessary to aid in future operation of passive treatment system.
- Continue environmental program (sampling/analysis/reporting)

Year 114 + (Open Pit Discharging)

- Open Pit Lake discharge treatment using passive biological system (North TMF Wetland)
- Pit water decant to North TMF Wetland during June through September. Pit water stored from October through May.
- Continue TMF seepage management as per years 32 – 114
- Continue environmental program (sampling/analysis/reporting)

## 5 – VEGETATION

The general objective for re-vegetation is to initiate the process for the return of the mine site to a condition which is similar to the existing natural vegetation. Existing vegetation in the area of the Casino Mine consists of black and white spruce in valleys and on lower slopes, with black spruce prevailing on wetter sites and white spruce on drier areas. In valley bottoms, sedge tussock fields are common. Alpine vegetation consists of scrub birch and stunted black spruce.

In general, the vegetation at the Casino site is typical of what is present throughout the Dawson Range ecosystem. No special or unique vegetation has been identified. There are minor occurrences of vegetation which is not populous, but is known to exist throughout the Dawson Range ecosystem.

The re-vegetation plan for the Casino mine is to control erosion of reclaimed areas and to initiate the transition to long-term or climax vegetation. Vegetation type will be adjusted for soil moisture, altitude and aspect to the sun.

Re-vegetation measures are expected to consist of:

- Placement of topsoil (taken from stockpiles developed during mine construction). In nutrient poor areas, vegetation establishment will be assisted by the use of early succession nitrogen fixers.
- Non-invasive species will be used, and use of native species will be promoted.
- Initial seeding of areas susceptible to erosion (slopes, etc.) with a native grass mix and a nurse crop to encourage rapid establishment.
- In areas less susceptible to erosion, a more natural approach to establish native species will be used, including woody species planting and local herb species establishment.
- The final phase of re-vegetation will be planting of spruce in patches or plugs to initiate the vegetation transition to climax vegetation.
- Techniques currently being tested by the Yukon Research Centre, and those used successfully at other mines in the Yukon, will also be incorporated where appropriate.

Details of the re-vegetation plan will be developed and updated throughout the mine life using the results from pilot plots and other testing at Casino, and in the larger research community.

### Topsoil for Re-vegetation

Soil material and quantities are covered in Section 3.8. Areas requiring topsoil for re-vegetation are shown on Figure 3.1.1. Topsoil for re-vegetation will be salvaged from areas to be disturbed during construction and mining. Topsoil salvaging will be selective to ensure that suitable material is available at closure. In general, coarse or sandy/gravelly soils are not ideal for re-vegetation, and preference for salvage will be on finer grain size soils. The focus of salvage will be on the upper 30 to 50 cm of soil which is best for reclamation purposes. Deeper soil may be salvaged and stockpiled separately to provide a sub-grade for topsoil in re-vegetation areas which are very rocky.

At a minimum, surface soils will be salvaged because they include the root layer of existing vegetation and generally have high organic matter content, which is best for reclamation.

It is expected that there will be some decay of the organic content of the topsoil over the mine life. An assessment of the topsoil will be made at the time of reclamation, and as necessary, nutrients/fertilizer will be added to assist in meeting the re-vegetation objectives.

All stockpiles will be constructed to ensure stability and control of erosion during the mine life. Stockpiles will be vegetated with a grass mixture to control erosion. Sediment settling ponds will be constructed as necessary downhill of stockpiles.

### **Wetland Vegetation**

Wetlands for passive water treatment are a key element of the closure plan. Wetlands will be constructed using a mix of vegetation typical of northern wetlands, with preference for both shallow and deep water. The exact mix and type of vegetation will be decided through the operation of an on-site study plot operated during the Operations phase of the project.

Wetland habitat may become an attractant to birds and various mammals, such as beaver and moose. The wetland will be designed to accommodate these species, while minimizing their effect on the wetlands functionality.

## 6 – WETLAND DESIGN

Water quality within the TMF Pond and Open Pit Lake will exceed CCME guidelines for certain parameters, and therefore this water should be treated prior to being released to the downstream receiving environment. The passive care solution proposed by CMC is the construction of engineered treatment wetlands that will treat discharge from the Open Pit Lake and the TMF Pond. The size and location of these wetlands, referred to as the North and South TMF wetlands, are presented on Figure 3.1-1. Design and support for the functionality and effectiveness of the wetlands is detailed in a memo prepared by Clear Coast Consultants, and is provided in Appendix A. This section provides a summary of the information in Appendix A.

### 6.1 GENERAL WETLAND DESIGN

Wetlands effect metal removal by a variety of mechanisms, including:

- Sorption and/or exchange onto organic matter [detritus]
- Formation of carbonates
- Association with iron and manganese oxides
- Reduction to non-mobile forms, and
- Formation of insoluble metal sulphides.

Plants play an essential role in forming and maintaining wetland sediments, but they do not accumulate metals and are generally recognized to account for less than 5% of their removal.

Several unique characteristics account for the unique capacity of wetlands to remove metals from solution in water:

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 and organic carbon 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 (CCC, 2013).

Both TMF wetlands will be designed as surface flow wetlands, which direct flow over the surface of the wetland as opposed to the subsurface. In these wetlands, metal removal occurs primarily at the sediment surface, rather than in plants or in the water column, and treatment performance depends on the total wetland surface. Surface flow wetlands are simple to design and operate, comparatively inexpensive, but they typically require a substantial surface area. Furthermore, wetland design must ensure the sluggish flow of water through the wetland, as mentioned above, and therefore the design



of the TMF wetlands will mitigate the potential for channelization of flow within the wetland and effective bypass of the treatment system. Finally, wetlands have been documented to still provide treatment even during the cold winter months. Only the South TMF Wetland is currently planned to treat water during the winter, and therefore the wetland will be configured to ensure treatment of these very small volumes during the winter.

## 6.2 NORTH TMF WETLAND WATER QUALITY AND SIZING

The water quality values used to design the North TMF Wetland were developed by Source Environmental Associates (SEA, 2013). The North TMF Wetland was designed to treat water discharged from the Open Pit after establishment of the pit lake approximately 95 years after closure of the Project. Discharge from the Open Pit is expected to occur at a controlled rate of 180 l/s; however, to ensure conservatism in design, the inflow rate to the North TMF Wetland was assumed to be roughly 20% greater, or 220 l/s.

The expected quality of the initial inflows to the North TMF Wetland, for key parameters exceeding CCME and treated by the wetland, is presented in Table 6.2-1. The wetland was designed to treat these parameters to CCME guideline levels upon discharge from the wetland. Consideration of the treatment rates per area of wetland for each of these parameters, predicts a wetland of 7 ha will be required to treat all of these parameters to CCME guideline. The North TMF Wetland has therefore been sized to have a surface area of 10 ha to include an additional measure of conservatism in the design.

**Table 6.2-1 North TMF Wetland Design Water Quality**

Parameter	Inflow		Outflow (CCME)	
	Concentration (mg/l)	Load (kg/day)	Concentration (mg/l)	Load (kg/day)
Cadmium	0.004	0.08	0.00008	0.002
Copper	0.37	7.0	0.004	0.08
Molybdenum	0.18	3.4	0.073	1.4
Uranium	0.062	1.2	0.015	0.29

## 6.3 SOUTH TMF WETLAND WATER QUALITY AND SIZING

The water quality values used to design the South TMF Wetland were also developed by Source Environmental Associates (SEA, 2013). The South TMF Wetland was designed to treat water in the TMF Pond prior to discharge down the spillway. The pond water quality is predicted to improve with time, with the poorest water quality occurring early in closure. The highest mean monthly discharge into the South TMF Wetland during this period is predicted to occur during May at a rate of approximately 0.44 m<sup>3</sup>/s (Kpf, 2013). The expected quality of the initial inflows to the South TMF Wetland, for key parameters exceeding CCME and treated by the wetland, is presented in Table 6.3-1. The wetland was designed to treat these parameters to CCME guideline levels upon discharge from the wetland. Consideration of the treatment rates per area of wetland for each of these parameters, predicts a wetland of 6 ha will be required to treat all of these parameters to CCME guidelines. However, it is understood that inflows will vary between years and the wetland will likely need to treat higher inflow rates. An assessment of flow variability during May within the baseline

hydrologic dataset for the TMF facility, which has a mean discharge of 0.45 m<sup>3</sup>/s (which is nearly identical to the modelled value), reveals that the 98<sup>th</sup> percentile monthly flow is 1.5 m<sup>3</sup>/s. To ensure treatment of higher inflows, the South TMF Wetland will therefore be sized to treat inflows of this magnitude to CCME. The result is a surface area for the South TMF Wetland of 20 ha.

**Table 6.3-1 South TMF Wetland Design Water Quality**

Parameter	Inflow		Outflow	
	Concentration (mg/l)	Load (kg/day)	Concentration (mg/l)	Load (kg/day)
Cadmium	0.0007	0.03	0.0001	0.003
Copper	0.139	5.28	0.004	0.15
Molybdenum	0.223	8.48	0.073	2.78
Uranium	0.063	2.40	0.015	0.58

## 7 – CLOSURE MAINTENANCE AND MONITORING

There are four sources of water that were identified as having the potential to adversely impact water quality to the receiving environment after closure of the project:

1. Pit water will require passive treatment via the North TMF wetland
2. HLF seepage is not predicted to require treatment
3. TMF pond water will require passive treatment via the South TMF Wetland, and
4. TMF seepage is not predicted to require treatment, as long as it is not discharged during the low flow winter months.

Ore stockpiles are processed, with residual material and foundations disposed of in the Open Pit, and therefore these are not considered a source after closure.

The closure strategy for the Casino Mine is to minimize the requirements for post-closure activities to the extent which is practical. No long-term active care is expected for the site; however, long-term passive care is expected. Passive care will consist of:

- Operation of valves to all gravity releases of water from the Open Pit, Winter Seepage Mitigation Pond, and if necessary the HLF and TMF. It is envisioned that the operation of the valves and control of flows could be done remotely, without any need for site presence.
- Periodic inspection and maintenance of:
  - Wetlands
  - TMF dam and spillway
  - Seepage mitigation pond dam and liner system
- A program of water quality monitoring at mine sources and in the receiving environment. This will include on-site and receiving environment water quality, metal accumulation in wetlands and plants, and other required activities as specified in the approved environmental management plans.

## **8 – TEMPORARY AND EARLY CLOSURE**

### **8.1 TEMPORARY CLOSURE**

In the event of a temporary closure of the mine, the following general activities will be conducted. A detailed temporary closure plan will be submitted in the Quartz Mining Licence and/or Water Licence application:

1. Pit sump pumps will be turned off and water allowed to accumulate in the pit.
2. Accumulation of water in the TMF will be monitored to ensure freeboard level is maintained at the dam. If necessary, surplus water will be pumped to the pit.
3. TMF seepage return pumping will be maintained.
4. Heap operations will continue with ongoing circulation of water onto the heap and processing of water in the recovery plant. Cyanide addition to the circulating water will be stopped. Surplus water will be processed in the Inco/SO<sub>2</sub> plant for cyanide destruction and pumped the pit.
5. Site power, security and personnel for the above activities will be maintained.
6. The mill will remain heated, but tanks and piping will be drained to the TMF.

### **8.2 EARLY CLOSURE**

In the event of an early closure of the mine, the following general activities will be conducted. A detailed plan for early closure will be submitted in the Quartz Mining Licence and/or Water Licence application:

1. Pit sump pumps will be turned off and water allowed to accumulate in the pit, all pit infrastructure will be removed, and the closure decant system will be installed.
2. Canadian Creek will re-directed to the open pit.
3. All LGO stockpiles will be processed as needed for TMF reclamation (minimum 1 m cover on all PAG rock) or relocated to the open pit for sub-aqueous disposal.
4. A spillway invert elevation, consistent the requirements for dam freeboard, will be determined. Any PAG higher than 1 m below the invert elevation will be relocated to provide 1 m cover.
5. Erosion protection will be placed on exposed sand areas of the dam.
6. Water in the TMF will be pumped to the pit, followed by construction of the wetlands.
7. The water retention pond for TMF seepage will be constructed.
8. Heap operations will continue with ongoing circulation of water onto the heap and processing of water in the recovery plant. Cyanide addition to the circulating water will be stopped. Once gold recovery ceases, water will be processed in the Inco/SO<sub>2</sub> plant for cyanide destruction and used to rinse the heap, drain-down water processed in the Inco/SO<sub>2</sub> plant for cyanide destruction and through the bio-reactor for selenium removal and then pumped the pit.
9. All infrastructure will be removed as per the closure plan.

## **9 – MINE RECLAMATION SECURITY**

CMC will provide financial security for the anticipated closure cost for the Casino Mine. An amount of \$125.9M at the end of mining has been included in the updated feasibility study (M3, Jan. 2013).

An updated estimate with supporting details will be provided in the Water Licence application. The updated estimate will present the ultimate cost at the end of mine life, and also the amount at select periods through the mine life. CMC will update the estimate and security provisions regularly throughout the mine life.

## **10 – SUMMARY**

This document provides a conceptual closure and reclamation plan for the proposed Casino Copper-Gold Project for review and approval by the Yukon Government to support the environmental assessment process. A conceptual design of closure and reclamation activities has been presented to meet the closure objectives of the project. The closure plan will be updated based on results of ongoing investigations and research.

Should there be any questions regarding the approach or conclusion of the report, please contact the undersigned.

Yours truly,

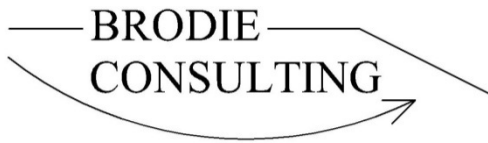
**Brodie Consulting Ltd.**

Lara Fletcher, P.Eng.

M. J. Brodie, P.Eng.

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## **APPENDIX A**

### **WETLAND WATER TREATMENT FOR THE CASINO PROJECT**

(Pages A-1 to A-54)



# Technical Memo

**To:** Kyle Terry  
**Cc:** Rina Freed, John Brodie, Doug Grimes, Rob Marsland  
**From:** André Sobolewski  
**Date:** December 13, 2013  
**Re:** Wetland Water Treatment for the Casino Project

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Several sources of contaminants at elevated concentrations have been identified in the modeling of water quality after closure of the Casino Mine. The contaminants originate primarily from three sources:

1. Heap draindown water,
2. Pit wall, and
3. Waste rock submerged within the TMF.

There are also minor contributions from other sources, but drainage from these sources commingles with the Pit Lake or TMF pond to result in elevated levels of certain parameters. Regardless of the source, these contaminants must be removed to protect the quality of drainage around the site.

All of the contaminants are predicted to be present at relatively low concentrations (i.e., < 5 mg/L), although flows are predicted to be relatively high (i.e., > 100 L/sec). In addition, mine drainage from all sources is predicted to be circumneutral. These conditions – low contaminant concentrations and neutral drainage – favour the use of passive treatment technologies. These technologies are potentially as effective as water treatment plants, but are not as burdensome, i.e., they do not require a permanent operator, power source, continual supply of reagent and removal of sludge. Only passive treatment systems will be considered in this closure plan, unless they are shown not to effectively mitigate contaminated drainage.

This document evaluates passive treatment options for the proposed Casino Mine. First, relevant background information will be presented to provide a context for this evaluation. The anticipated flow rates and chemistry from each major source will then be reviewed separately. Finally, the potential for a passive treatment system to remove constituents from each source will be assessed, given all the conditions anticipated at the site, including during the winter.

Treatment wetlands will be considered for treatment of Pit Lake TMF pond drainage. The assessment of the North TMF Wetland will be far more comprehensive than that of the South TMF Wetland, since both wetlands will treat the same contaminants. It is understood that the process chemistry, case studies and sizing equations used to design the North TMF Wetland are equally applicable to the South TMF Wetland and they do not need to be repeated.

The heap draindown will be directed to the Open Pit, where it will be diluted with precipitation and other water draining into the pit. Ostensibly, draindown water should not require separate treatment because all water that eventually discharges from the Pit will be treated in the North Wetland. However, mercury and selenium, which are found predominantly in heap draindown water, as still at elevated values within the Open Pit discharge, even after dilution in the Pit Lake. These contaminants are generally contraindicated for wetland treatment because of their potential to enter the food chain. For this reason, draindown water may require additional treatment targeted at selenium and mercury prior to it entering the North TMF Wetland.

Different options for addressing elevated levels of mercury and selenium are available. For example, accounting for the precipitation of selenium with sodium sulfide during the gold recovery SART process could reduce selenium levels in the re-circulated HLF water, and therefore also the draindown water<sup>2</sup>. It may also be possible to treat mercury and selenium during the operation of the heap by adding unit processes after the gold recovery circuit. For example, the Key Lake Mine added a chemical process for selenium removal to its existing uranium extraction and water treatment circuit<sup>1</sup>. Apparently, their treatment process is not based on the ferrihydride precipitation process, which is an accepted, well documented process (Sobolewski, 2005). Similarly, a number of chemical processes for removal of mercury to very low concentrations could be applied, such as applying proprietary reagents such as TMT-15 or Nalco 8068. However, the passive treatment option currently selected for closure design of the Casino Project is a bioreactor. Mercury and selenium would both be removed from HLF draindown prior to it entering the open pit, which would protect the North TMF Wetland from being exposed to their elevated concentrations. Selection of this mitigation treatment option is decidedly conservative from a closure design and costing perspective, and was selected for this reason along with the effectiveness and passive nature of the treatment.

The primary documents used in this evaluation are:

1. The Casino Project - YESAB Water Balance Report prepared by Knight Piésold, dated December 2013.
2. The Casino Project – Water Quality Prediction report prepared by Source Environmental Associates, dated December 2013

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<sup>1</sup> Personal Observation.

<sup>2</sup> Email correspondence with John Marsden.

3. The Casino Project – Geochemical Source Term Development prepared by Lorax, dated December 2013.

In this evaluation, I will rely on my experience working in The Yukon to assess feasible options and avert options that are likely unfeasible at this site.

**RELEVANT SITE CHARACTERISTICS**

The Casino Project property is located in the west central Yukon, in the Dawson Range Mountains, approximately 300 km northwest of Whitehorse. Typical of this part of the Yukon, the range comprises ridges and hills of moderate elevation, as well as deeply incised valleys. The site drains primarily into Casino Creek, which flows to the Yukon River via Dip Creek, the Klotassin River and the White River. The Open Pit also extends into the Canadian Creek watershed, which flows to the Yukon River approximately 16 km north of the project site.

The climate is generally cold, with a mean annual temperature estimated at -3.2°C, and with minimum and maximum mean monthly temperatures of -18.0°C and 11.4°C in January and July, respectively. Cold, sub-zero temperatures dominate from approximately October through May, which restricts the period of growth and strong biological activity to four months. Thus, biologically-based treatment systems that cannot operate during the winter are not acceptable at this site.

The site hydrology is largely governed by spring snowmelt and summertime precipitation, as runoff decreases considerably during the cold winter months when temperatures are well below zero. In the post-closure scenario being considered, the wetland hydrology will be determined almost entirely by the discharge from the pit lake, since it will be the main source of water into the wetland.

The elevation of the study area ranges from approximately 650 masl at Dip Creek to 1400 masl near the proposed open pit. The overall site topography is hilly, with few stretches of flat or gently-sloping ground in the project area that could accommodate a large treatment wetland (Figure 1). The valley bottoms below the project area also do not appear to offer large flat areas.

The site is presently underlain by discontinuous permafrost, primarily located in the valley bottom and North-facing slopes. It is not clear if this discontinuous permafrost will remain as climate warms near the end of mine life. However, permafrost is not expected to prevent the development of a treatment wetland, since it was not a significant factor at more northerly sites (Minto Landing, Keno Hill). Furthermore, the wetlands will be constructed on submerged tailings within the TMF pond, and it is not anticipated that permafrost would develop in this environment.

The site vegetation is dominated by muskeg and Black Spruce. Stands of sedges (e.g., *Carex aquatilis* or *rostrata*) will be sought near the project area to vegetate the treatment wetlands. These stands will be used as donor sites, though it will be necessary to propagate the plants to obtain sufficient numbers.

There are no large wetlands in the project area that would attract wildlife associated with these ecosystems. However, the presence of the new pit lake, the TMF pond, and the construction of treatment wetlands (Figure 1) is expected to attract such wildlife<sup>2</sup> after mine closure. This must be accounted for in the wetland design:

1. The design of the wetlands must resist damage that would compromise its integrity from resident or roaming wildlife (beavers, moose, muskrats, etc).
2. Any toxic metal retained in the wetlands must not be taken up by plants and allowed to accumulate in the food chain to harm either resident or transient wildlife.

As indicated, the above site characteristics have implications for the design of a treatment wetland. These implications will be discussed in later sections.

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<sup>2</sup> Some of this wildlife, such as beavers or muskrats, may become permanent residents of these new ecosystems, while others, such as snow geese or other migratory species, may become transient visitors. In the latter case, it is possible that they will raise their chicks in the wetlands, which may accentuate any risk from contaminant transfer in the food-chain.



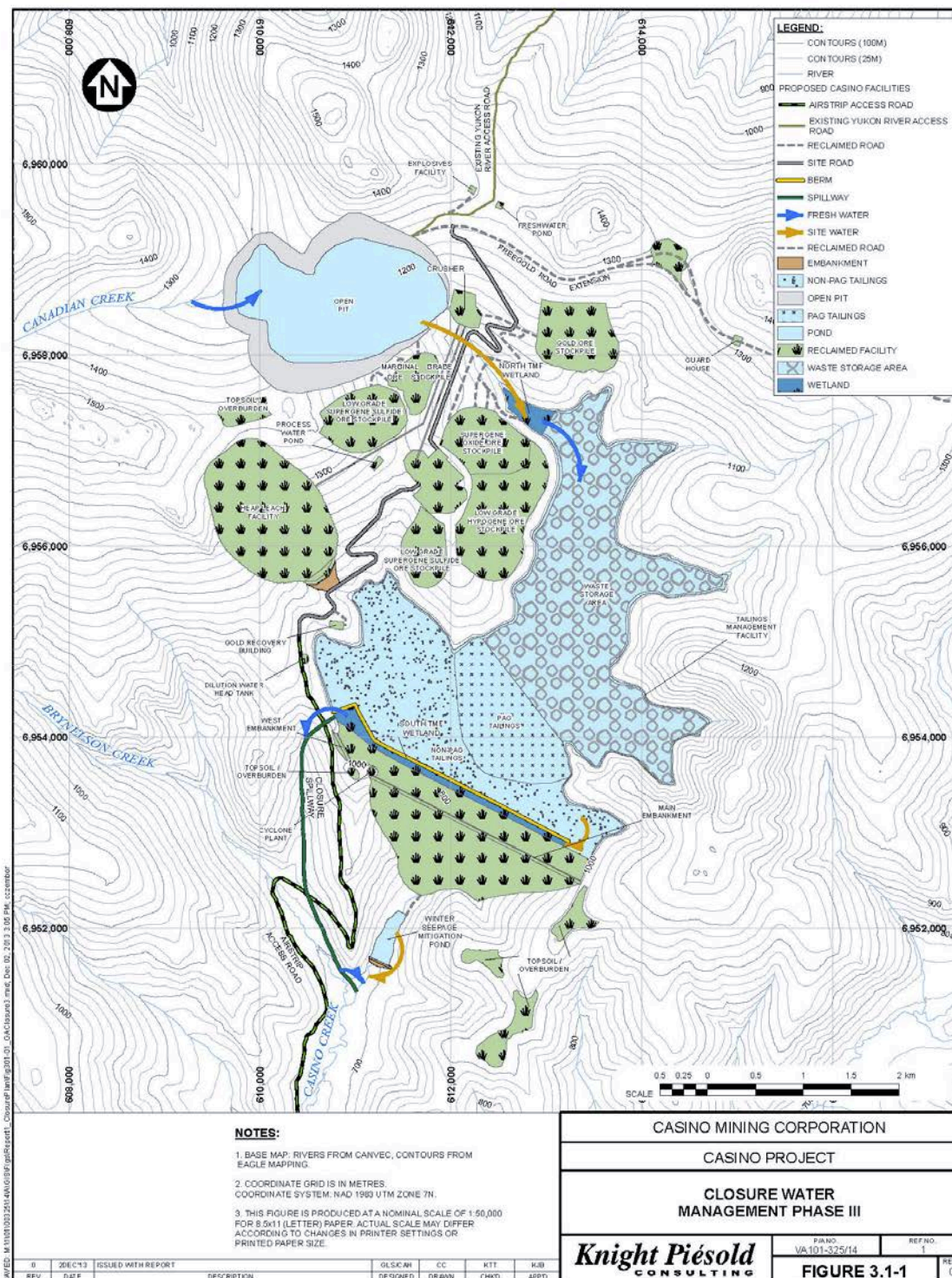


Figure 1. Anticipated site plan at closure.  
Note the area proposed (green shade) for treatment wetlands.

## **RATIONALE FOR USING TREATMENT WETLANDS AT CLOSURE**

The appeal of using treatment wetlands at mine closure is two-fold:

1. There are numerous reports of natural wetlands that remove metals from natural or mine waters. Moreover, since these ecosystems appear to be healthy, it suggests that retained metals do not harm them.
2. Treatment by wetlands occurs naturally and does not require an operator, power or reagent addition.

This information suggests that treatment wetlands could provide low-cost “walk-away” treatment of mines at closure, whereby metals are continually removed without the need for human intervention. Unfortunately, there are only a few examples where wetlands treating mine drainage were truly “walk-away” solutions (e.g., the Snake Island fen downstream of the Cluff Lake mine). Most treatment wetlands require some form of passive management, such as periodic maintenance. Still, where applicable, they represent one of the most cost-effective solutions, when compared with other alternatives.

Exploration geologists have long known that wetlands retain metals, but their intent was to locate ore bodies, not to treat mine drainage. The use of wetlands to treat mine drainage goes back to the late 1970’s/early 1980’s when scientists documented the improvement by wetlands of contaminated drainage from abandoned coal mines in Eastern Appalachia (See Sobolewski, 1996a). Although the concept of using natural systems to remove metals from mine water was relatively new, it was already well understood by exploration geologists. They recorded numerous examples of cupriferous or uraniferous bogs, natural systems that retain metals in their sediments, and used them to locate ore bodies. The early designs of treatment wetlands were haphazard, but a 1994 publication by the former US Bureau of Mines established a scientific basis for their design, as applied to coal mine drainage.

Similar efforts were made in the 1980’s and 90’s for base and precious metal mines in Western North America, encouraged by findings of natural wetlands that treated drainage at the operating or abandoned Carbonate (Montana), St Kevin Gulch (Colorado), Birchtree (Manitoba), Cluff Lake (Saskatchewan), Star Lake (Saskatchewan), Silver Queen (British Columbia) and Keno Hill Mines (Yukon). Investigations of constructed wetlands at the Big Five Tunnel (Colorado), Dunka (Minnesota) Bell Copper (British Columbia) and Keno Hill Mines established four important points:

1. Metal uptake by plants is minimal and does not account for significant removal. Metal levels in plant tissues are comparable with those of plants sampled from nearby control (i.e., unimpacted) areas.
2. Sulphate is reduced in wetland sediments into hydrogen sulphide, which reacts with dissolved metals to form highly-insoluble metal sulphides. Other metals (copper, uranium) form very strong bonds with organic matter. Metals retained within sediments will “age” into unextractable forms.
3. Metal removal has been documented during the winter in very cold climates.

4. Although wetlands can treat mildly acidic mine water ( $\text{pH} > 5$ ), they are ineffective in treating more acidic drainage ( $\text{pH} < 4$ ).

These characteristics reinforce the view that treatment wetlands are among the best options for treating mine drainage at closure. In a later section, I will review examples of wetlands that treat drainage contaminated with the metals expected in the discharge from the closed Casino Mine.



## HEAP LEACH FACILITY –BIOREACTOR DESIGN

A gold extraction heap leach facility (HLF) will be constructed and operated as part of the Casino Project. When gold extraction is complete, the spent heap will be rinsed to destroy cyanide residues. Thereafter, the spent heap will be drained down and reclaimed. The draindown water is predicted to contain several metals and other constituents at elevated concentrations. The characteristics of the heap draindown and design of the bioreactor are discussed in this section.

### PREDICTED DRAINDOWN CHEMISTRY

Knight Piésold Ltd. (KP) predicted flow rates for the heap draindown, based on stored water volumes and meteorological characteristics of the site. During the five year draindown period flows from the HLF are predicted to range from **20-63 L/sec** (Table 1 and Figure 2). Thereafter, flows are expected to decrease to 0-24 L/sec.

**Table 1. Monthly Flow Rates for Heap Draindown<sup>1</sup>**

Flow (L/sec)	Minimum	Maximum	Average
Five year draindown period	20	63	33
Long term flows <sup>2</sup>	0	44	13

<sup>1</sup> Provide by KP and sourced from the Casino YESAB Water Balance Model.

<sup>2</sup> Includes both surface runoff and seepage toe discharge

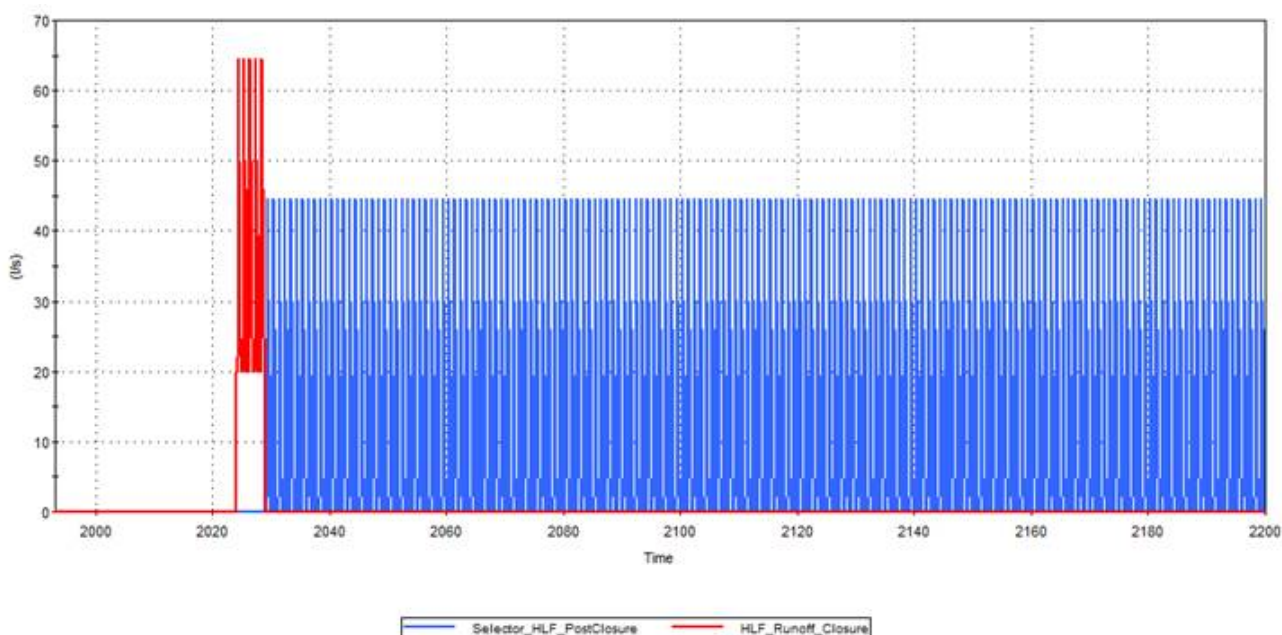


Figure 2. Predicted flow rates from spent heap.

Lorax Environmental predicted the source term used in determining average the water quality of the heap draindown. HLF source terms are provided for three periods:



immediately after closure during HLF draindown, during the first 10 years following draindown, and during the long-term. These predictions are presented in Table 2.

Table 2 shows that concentrations of arsenic, cyanide (Total and WAD), cadmium, copper, mercury, molybdenum, silver, selenium, uranium and zinc are elevated relative to CCME and other relevant Guidelines for this site. Most of these exceedances are for the initial draindown, but some contaminants persist at elevated concentrations for ten years afterwards, including arsenic, cadmium, copper, molybdenum, selenium, uranium and zinc.

Note that comparing these exceedances to Guideline levels is arbitrary, because the heap draindown will be conveyed to the pit lake, rather than being discharged directly to the TMF pond. Nonetheless, these levels are useful guides to identify potential contaminants.

**Table 2. Predicted Heap Draindown Water Quality<sup>1</sup>**

Parameter	Initial Draindown	First ten years	Long Term Draindown	Water Quality Guidelines
pH	8.71	7.7	7.7	6.5 – 9.0
Sulphate (SO <sub>4</sub> )	1,920	2,100	424	-
Arsenic (As)	0.036	0.036	0.036	0.005
Cyanide (Total)	10.7	0.2	<0.01	1
Cyanide (WAD)	5	0.2	<0.01	0.005
Cadmium (Cd)	0.00843	0.00498	0.000275	0.000055 <sup>2</sup>
Calcium (Ca)	666	756	532	-
Cobalt (Co)	5.69	2.55	0.489	-
Copper (Cu)	2.75	0.016	0.00112	0.004 <sup>2</sup>
Iron (Fe)	9.46	0.00403	0.00403	-
Mercury (Hg)	0.0183	0.00002	0.00002	0.000026
Molybdenum (Mo)	4.18	4.18	0.939	0.073
Lead (Pb)	0.00147	0.000279	0.000279	0.0071 <sup>2</sup>
Nickel (Ni)	0.241	0.0934	0.00727	0.150
Silver (Ag)	0.0534	0.000778	0.000622	0.0001
Selenium	0.4	0.227	0.0975	0.001
Uranium (U)	0.0018	0.627	0.173	0.015
Zinc (Zn)	0.0716	0.389	0.221	0.030

<sup>1</sup> Taken from Lorax Environmental: Casino Geochemical Source Term Report (December 12, 2013). All values in mg/L, except for pH (expressed in standard units).

<sup>2</sup> Value corrected for hardness, assumed to be 180 mg/L of CaCO<sub>3</sub>.

Arsenic and zinc will become so diluted in the pit lake that their concentrations will decrease to below CCME Guidelines. However, cadmium, copper, mercury, molybdenum,

selenium and uranium are predicted to still exceed CCME Guidelines in the pit lake discharge. A bioreactor could treat all of these metals; however design of the heap draindown bioreactor will focus on mercury and selenium as they can potentially enter the food chain if retained in wetlands.

Several of the above metals are likely to exist as cyanide complexes in the HLF draindown water, reporting as Total and/or Weak-Acid Dissociable (WAD)-CN. These include copper, cobalt, iron, mercury, nickel, silver and possibly cadmium and zinc. Among those, cadmium, copper, silver and zinc form WAD-CN complexes, whereas cobalt, iron, mercury and nickel form strong cyanide complexes (T.W. Higgs Associates, 1992). The former complexes will readily dissociate as draindown pH decreases towards neutrality, whereas the latter will remain stable. However, the strong complexes of cobalt-cyanide and iron-cyanide are photo-sensitive and will dissociate in sunlight (Leduc et. al., 1982). These characteristics will be considered in selecting treatment options for heap draindown.

### FEED CHEMISTRY AND DESIGN CRITERIA FOR HEAP DRAINDOWN BIOREACTOR

The design of the heap draindown bioreactor will focus on the treatment of selenium and mercury because of their potential to exert ecotoxic effects in the North TMF Wetland. The draindown concentrations provided in Table 2, along with the predicted draindown flows, will provide the design basis for a treatment system.

Water from the HLF is only directed to the Open Pit during draindown, and then eventually onward to the North TMF Wetland, and therefore treatment will only be required during the 5-year HLF draindown phase (Project Years 23-28). During that time, the treatment system should be designed to treat the worst case flows and contaminant concentrations.

Average and maximum monthly flows from the spent heap during this period are predicted to be 33 and 63 L/sec, respectively. Therefore, the design flows for this treatment system are set at **63 L/sec**.

Worst case concentrations of selenium and mercury in the spent heap draindown are shown in Table 3. Their concentrations should be decreased by 90% so that their levels in the pit lake, after dilution, do not impair the function of the planned treatment wetland receiving the pit lake drainage.

**Table 3. Worst Case Predicted Concentrations of Heap Draindown Contaminants.**

Constituent	Worst Case Concentration <sup>1</sup>	Target Discharge Concentrations <sup>1</sup>
Copper	2.75	0.028
Cyanide	10.7	1.1
Mercury	0.0183	0.0002
Silver	0.0534	0.0005
Selenium	0.4	0.004
Uranium	0.627	0.0063

<sup>1</sup> All concentrations in mg/L

The above table does not provide the nitrate concentrations in the heap draindown, nor were these predicted by Lorax. However, nitrate is an important constituent to consider when designing an anaerobic bioreactor, because its complete removal is necessary before metal sulphides, selenium and uranium can be removed.

In the absence of predicted concentrations, it will be assumed that nitrate concentrations in the heap draindown will be 100 mg/L as  $\text{NO}_3^-$ . By comparison, nitrate concentrations in the spent heap draindown at Brewery Creek ranged from 0-40 mg/L. At Goldcorp's former Wharf Resources Mine, in Leads, SD, mine water collected from barren rock contained approximately 30 ppm nitrate. At the former Zortman-Landusky gold mine, in Montana, the acidic water collected from leach pads contained approximately 200 ppm nitrate. Draindown from spent heaps in Nevada can contain much higher nitrate concentrations (0.5-1.0 g/L), but their concentrations are much higher than at heaps operated in cold climates due to the concentrating effect of water evaporation. Taking this information together, nitrate concentrations of 100 mg/L in the heap draindown is a reasonable assumption.

## PROCESS CHEMISTRY

The contaminants shown in Table 3 are solubilized under oxidizing conditions and made insoluble under reducing conditions. Therefore, treatment involves conveying heap draindown in a reducing environment and allowing the contaminants time to react and form insoluble compounds. In a bioreactor, this is accomplished by circulating mine water through organic material that decomposes in the absence of oxygen. In the presence of sulphate, sulphate-reducing bacteria (SRB) utilize organic matter and generate hydrogen sulphide, which reacts with most metals to form highly insoluble compounds.

Table 4 shows the solubility products<sup>3</sup> for copper, mercury and silver. In every case, the solubility products indicate that their metal sulphides are highly insoluble. Thus, metals in heap draindown water forced to flow through a reducing environment containing hydrogen sulphide will react to form insoluble compounds and will be retained in that environment.

*Table 4. Solubility products:  $\log K_{sp}$  of some metal sulphides at 25 °C.*

Metal Sulphide	$\log K_{sp}$	Metal Sulphide	$\log K_{sp}$
$\text{Ag}_2\text{S}$	-50.1	$\text{HgS}$	-52.7
$\text{CdS}$	-25.8	$\text{MnS}$	-10.5
$\text{CoS}$	-21.3	$\text{MoS}_2$	-63.5
$\text{CuS}$	-36.1	$\text{NiS}$	-19.4
$\text{Cu}_2\text{S}$	-47.7	$\text{PbS}$	-27.5
$\text{FeS}$	-18.1	$\text{ZnS}$	-24.7

Table taken from Jackson, 1986. Value for  $\text{MoS}_2$  is from Kaback and Runnells, 1980.

<sup>3</sup> The solubility product  $K_{sp}$  is essentially defined as the product of soluble species concentration over that of insoluble species, at equilibrium. Metal sulphides with  $\log K_{sp}$  values below -25 are highly insoluble, whereas metal sulphides with values above -20 are moderately soluble.

Metals like mercury or silver will probably enter the treatment system as metal-cyanide complexes. The formation of metal sulphides is thermodynamically favoured, but the exchange from cyanide complexes to sulphides will be kinetically constrained. Therefore, longer retention times will be needed than if the metals were simply the dissolved species in solution.

Selenium process chemistry is somewhat more complex. In oxic environments, selenium exists as the fully-oxidized, non-reactive selenate ( $\text{SeO}_4^{2-}$ ) (See Figure 3). In anoxic environments, it exists as the partially-reduced, reactive selenite ( $\text{SeO}_3^{2-}$ ). In the environment of a rinsed heap, selenium could exist in either form, but is more likely to be present as the non-reactive selenate.

Treatment in a bioreactor consists of reducing selenate and selenite into the insoluble elemental selenium (Sobolewski, 2005). This reaction is carried by selenate-reducing bacteria as well as sulphate-reducing bacteria. In practice, this is achieved by promoting conditions that favour sulphate-reducing bacteria, which also produce hydrogen sulphide and remove the above metals.

Uranium geochemistry is the most complex, but its treatment turns out to be much the same as above. In the environment of the heap, the oxidized uranyl anion [U(VI)] is likely to exist predominantly as the mobile uranyl-carbonate complex, with minor amounts of uranyl-fluoride and uranyl-phosphate complexes). The actual species depends on ambient pH (See Figure 3). When these complexes enter the circumneutral, reducing environment created by SRB in a bioreactor, they breakdown into the highly insoluble uraninite ( $\text{UO}_2$ ) (Luo et. al., 2007).

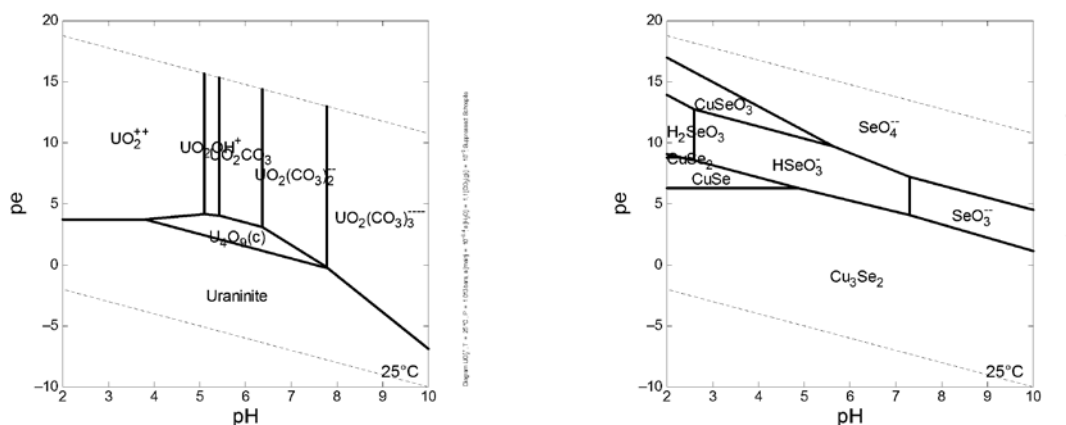


Figure 3 – Stability field diagrams for uranium and selenium. Graciously provided by George Breit, USGS. Conditions for calculations:  $T = 25^\circ\text{C}$ ,  $a[\text{H}_2\text{O}] = 1$ ,  $f[\text{CO}_2(\text{g})] = 10^{-2}$ ,  $a[\text{Cu}^{2+}] = 10^{-7}$ , suppressed Se(black)

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. Copper, mercury or silver may react with hydrogen sulphide to form insoluble compounds, but at low concentrations, a portion will remain suspended as colloidal compounds. Their removal

from solution as sulphides to very low concentrations requires that they contact adsorptive surfaces.

Finally, nitrate is easily removed biologically in anoxic environments. In the proposed bioreactor treatment, three processes can be active to remove nitrate from heap draindown:

1. Bacterial denitrification will convert nitrate to dinitrogen gas, while oxidizing organic matter in the bioreactor matrix,
2. Bacterial denitrification will convert nitrate to dinitrogen gas at the expense of liquid organic carbon (ethanol, ethylene glycol) that may be supplied to the bioreactor,
3. Bacterial denitrification will convert nitrate to dinitrogen gas at the expense of elemental sulphur that will be incorporated in the matrix of the bioreactor.

While the first process is relatively temperature-sensitive, the last two are relatively insensate and will occur as long as water can flow through the system. These processes will be incorporated in the overall design of the bioreactor.

### CONCEPTUAL BIOREACTOR DESIGN FOR TREATMENT OF HEAP DRAINDOWN

A passive bioreactor is proposed to treat the heap draindown during active closure. Passive bioreactors use a simple flow-through design, with a solid reactive mixture acting as a source of carbon for the bacteria and as a substrate for microbial attachment and metal sulfide precipitation.

The proposed bioreactor draws from a concept developed at Homestake Mining Company's Santa Fe Mine, in Mineral County, Nevada. At that mine, the "Biopass system" was developed in the early 1990's for treatment of drainage from a reclaimed cyanide heap leach facility (Cellan et. al., 1997). This system relies on the flow of mine water through a pile of organic matter undergoing anaerobic decay (Figure 4). It is designed to treat weak acid dissociable (WAD) cyanide (CN), nitrate (NO<sub>3</sub>), mercury (Hg), and selenium (Se).

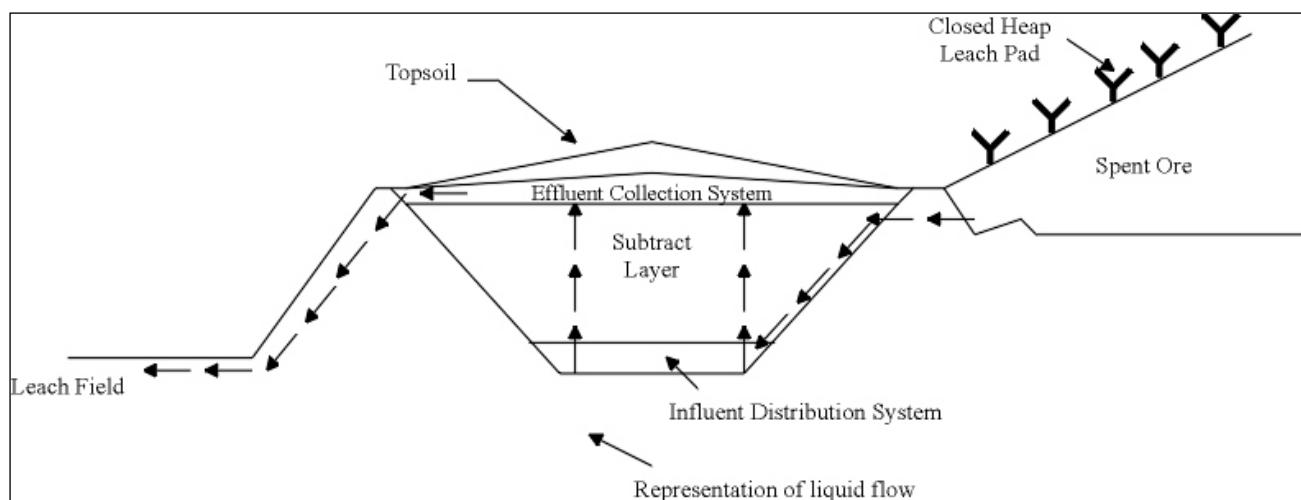


Figure 4. Schematic diagram of the Biopass system.



The Biopass system is constructed in an emptied double geomembrane-lined solution pond (Figure 5) and consists of, from the bottom upward: a seepage collection (influent) layer comprised of gravel and perforated pipe, a substrate layer comprised of a spent ore gravel and composted cow manure mixture, an effluent collection layer comprised of gravel and perforated pipe, a geotextile cushion, a geomembrane liner, and vegetative soil cover (Figure 6). Treated solution flows by gravity through a buried pipeline from the effluent layer to a leach field where it is aerobically treated. SRB in the substrate layer generate hydrogen sulphide, which forms insoluble mercury sulphides and reduce selenium to insoluble elemental selenium. They also decrease sulphate concentrations in the effluent. Other microorganisms biodegrade cyanide and remove nitrate from the water.



Figure 5. Placement of substrate. Note thick layer of organic matter.



Figure 6. Placement of substrate nearly complete.

The challenges associated with design of a bioreactor included:

- characterizing expected draindown quantity and quality;
- establishing organic substrate mass and treatment retention times;
- devising and placing the substrate layer to minimize consolidation and maintain permeability;
- developing *in situ* field construction quality-assurance procedures;
- minimizing construction/installation difficulties.

The total mass of organic matter required in this system was calculated on the basis of the anticipated sulphate loads, reflecting the fact that sulphate is the dominant anion. A predicted sulphate load is calculated using the average annual flow rates, summed over a 20 year period with two 100 year, 24-hour storm events, and an average sulphate concentration of 2,000 mg/l. The mass of organic substrate is determined assuming a conservative ratio of volatile suspended solids that will be consumed per mass of sulphate.

The selection and placement of the substrate was designed to allow flow of the anticipated peak discharge and provide a minimum hydraulic conductivity over the life of the system, accounting for substrate comminution and consolidation. The correct substrate mix was

selected by evaluating substrate combinations in laboratory studies. Careful application of the selected substrate mix during the construction of the system was ensured by maintaining a rigorous Quality Assurance/Quality Control program.

In its first year of operation, the Biopass system reduced the concentration of WAD cyanide by 83%, sulfate by 25%, nitrate by 91%, mercury by 98%, and selenium by 82%. The Biopass system is operating at approximately a 20-day retention time and a flow rate of approximately 0.44 liters per second (7 GPM).

The Biopass system was one of the early bioreactor designs for anaerobic treatment of metals. Since then, improvements have been made to their designs so that full treatment can be provided with lower retention times (and a smaller footprint). Improvements include:

- Selection of matrix material that minimize its plugging,
- Incorporation of sulphur prills that enhance the rate of hydrogen sulphide generation (resulting in quicker treatment) and that allow its formation at lower temperatures
- Provision of liquid carbon (e.g., methanol, ethanol, or ethylene glycol) to enhance the rate of hydrogen sulphide generation and allow operation at cold temperatures.

The incorporation of sulphur prills into an organic matrix (such as compost or manure) draws from findings with tank-based bioreactors that sulphide production at the expense of elemental sulphur is more rapid than from dissolved sulphate. This concept was validated in a passive bioreactor operated for 10 years at the Tulsequah Chief Mine (Sobolewski, 2010). This mine produced strongly acidic drainage that contained very elevated concentrations of copper and zinc. These metals were removed efficiently by precipitation as metal sulphides in the anaerobic component of this treatment system, in which sulphur prills were incorporated in the organic matrix.

That same bioreactor also demonstrated the benefits of utilizing liquid organic carbon instead of relying on the decomposition of organic matter, for cold temperature operation<sup>4</sup>. A similar design was demonstrated on a pilot-scale for the treatment of selenium from coal mine drainage. Selenium removal rates of 95% were documented even when water temperature reached 2 °C during winter months in Alberta (Sobolewski, 2010).

Current designs reflect these and other improvements in treatment performance. Minimum contact times in modern bioreactors generally range from eight to 48 hours (ITRC 2010). For example, the 3,000 m<sup>3</sup> West Fork bioreactor treats 1,200 gpm mine effluent containing lead at 0.4 mg/L and zinc at 0.36 mg/L down to <0.05 mg/L, on a nominal retention time<sup>5</sup> of 12 hours (Gusek et. al., 2000).

The treatment objectives for a bioreactor at Casino are to:

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<sup>4</sup> Average water temperature at Tulsequah was 6-7 °C.

<sup>5</sup> The retention time is referred to as nominal because it is calculated from the bioreactor volume and flows. The actual retention time is determined using tracer studies, whereby the transit time of mine water through the reactor is measured directly.

- Remove nitrate to  $<0.1$  mg/L
- Remove metals and selenium by 90%

Since most of the metals<sup>6</sup> are complexed with cyanide, any pre-treatment that can breakdown the complexes should be applied first. This means decreasing the pH of heap draindown to promote dissociation of WAD-CN complexes and exposing it to sunlight so that photo-sensitive complexes can breakdown. In practice, this is achieved by sending heap draindown in a shallow storage pond with a 24 hour retention time. Water pH and cyanide concentrations gradually decrease during the heap detox, creating favourable conditions for the dissociation of metal-cyanide complexes in a shallow open pond. In addition, suspended solids will be retained in this pond, an essential pre-treatment for a bioreactor.

The discharge from this pond will be directed to a two-stage bioreactor. The first stage is designed to remove oxygen and nitrate from draindown and create anaerobic conditions. This will be accomplished by incorporating sulphur prills in the bioreactor matrix. Elemental sulphur will be oxidized both by oxygen and by nitrate, the latter through sulphur-based denitrification. The products of these reactions are sulphate, some proton acidity and dinitrogen gas ( $N_2$ ). Proton acidity will easily be buffered by solution alkalinity. Escape of dinitrogen gas formed in the bioreactor will be promoted by utilizing a relatively porous medium (gravel-sized rock, as opposed to sand) and designing an escape pathway at the top of the bioreactor.

Water from the first bioreactor stage – now free of oxygen and nitrate, and with a low ORP – will be sent to a second stage that promotes SRB activity. This will be done by filling the bioreactor with a matrix that comprises a mix of organic and inorganic bulk material, sulphur prills, and by feeding liquid organic carbon. In this environment, where high  $H_2S$  concentrations are produced, copper, mercury and silver will form highly insoluble metal sulphides, even if present as cyanide complexes. In addition, selenium will be reduced to elemental selenium and uranium will be reduced to uraninite. Depending on the ambient ORP, selenium may even be reduced to selenide ( $HSe^-$ ), which forms an extremely stable, highly insoluble compound with mercury ( $HgSe$ ).

Like the Biopass system, the proposed bioreactor will receive water in an upflow configuration and will be covered. This will prevent animals from burrowing into the matrix and remobilize metals retained within the matrix. As with that system, the matrix composition will be formulated to prevent plugging from compaction or accumulation of solids.

A final, polishing basin, or aeration channel, will receive the bioreactor discharge and remove hydrogen sulphide by promoting its oxidation to elemental sulphur. This will eliminate the health and safety hazards from hydrogen sulphide offgasing to an area that exposes mine personnel.

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<sup>6</sup> Selenium is a metalloid, not a metal, and does not form soluble complexes with cyanide. However, it may react with cyanide to form  $SeCN$ , which is chemically equivalent to thiocyanate.



The dimensions of this treatment system are estimated to determine if, on first approximation, it can be realistically constructed near the spent heap at closure. It must be born in mind that these dimensions and volume are only estimates, based on current predictions of heap draindown flows and chemistry. Nonetheless, they are useful in judging if the proposed treatment concept can realistically be implemented at this site.

The shallow pond that removed suspended solids and promotes dissociation of photosensitive metal-cyanide complexes should have an 18 hour retention time. Given design flows of 63 L/sec (5443.2 m<sup>3</sup>/day), a 4,084 m<sup>3</sup> pond will be adequate for this purpose.

The bioreactor dimensions are estimated as follows.

The volume for the first, denitrification stage of the bioreactor is determined by the volumetric rate of denitrification. In a previous study (partly reported in Sobolewski, 2010), a pilot-scale, 30 L bioreactor fed mine water containing 15-20 mg/L NO<sub>3</sub><sup>-</sup> completely removed nitrate in 1.5 hours through sulphur-based denitrification. Ambient water temperature in this system was 21-22 °C, whereas draindown temperature is expected to be 4-6 °C during treatment. Applying a Q<sub>10</sub> temperature coefficient of 2.5 (typical for most biological processes) to the above results, a retention time of  $1.5 \times 3.75 \times 5 = 28.125$  hours<sup>7</sup> would be expected at 5 °C or rounding up, 28 hours. Given flows of 63 L/sec, a volume estimated at 18,144 m<sup>3</sup> would be required to remove nitrate from heap draindown<sup>8</sup>.

Although the Biopass bioreactor resembles most the proposed Casino bioreactor in its application, the West Fork bioreactor is actually a better analog for sizing purposes because the flow rates between these two systems are similar (63 L/s for Casino vs 75 L/s for West Fork). Its nominal retention time of 12 hours provides a basis for sizing the Casino bioreactor. It suggests that a retention time of 24-30 hours will be sufficient to remove dissolved metals from heap draindown.

An additional factor to consider for the Casino bioreactor is that most metals are present as cyanide complexes. The time required to dissociate these complexes must be added to the time required to form insoluble sulphides and precipitate within the bioreactor. Higgs and co-workers report that the silver-cyanide complex takes less than 48 hours to equilibrate in solution (Higgs et. al., 1992), suggesting that an additional 48 hours should be added to the bioreactor retention time<sup>9</sup>. Taken together, this suggests that a bioreactor with a 60 hour retention time will remove metals and selenium from heap draindown.

Combining the 28 hour retention time for nitrate removal and 60 hour retention time for metal removal, a bioreactor with an 88 hour retention time is expected to treat heap

<sup>7</sup> The equation is: Retention Time (RT) @ 21 °C x Q<sub>10</sub> correction factor for 5 °C x estimated nitrate concentration = bioreactor RT

<sup>8</sup> Assuming that porewater occupies 35% of the total bioreactor volume.

<sup>9</sup> The assumption of reversible dissociation is very simplified in this argument because it depends on several factors, notable the ambient cyanide, metal and sulphide concentrations, temperature, solution pH and other factors. Laboratory studies are required to determine the actual removal rates inside a bioreactor.

draindown and meet the abovementioned treatment objectives. **Assuming a design flow of 63 L/s, the nominal volume for this bioreactor is estimated at 57,024 m<sup>3</sup>.** Even if this estimated volume is too small, the Event Pond below the heap has a storage capacity of 74,000 m<sup>3</sup>, which represents a retention time of approximately 130 hours and is more than adequate for this purpose.

This bioreactor will be preceded by a shallow 5,000 m<sup>3</sup> pond and followed by a smaller channel to aerate water and remove hydrogen sulphide from the bioreactor effluent. Figure 1 shows that there is sufficient room below the closed heap to accommodate for such a treatment system.

This system could be operated in two distinct modes. During active closure, when contaminant concentrations in draindown are elevated, the two-stage system will receive liquid organic carbon to enhance sulphate reduction and metal removal. Provision of liquid organic carbon would be discontinued after this period, when contaminant levels have decreased.

Once active closure is finished, this system could be operated for long-term passive treatment if required, or sealed outright.

The above pond and bioreactor volumes are estimates based on the performance of existing systems. As predictions of draindown flow and chemistry improve, it will be necessary to revise these estimates. In addition, bench- and pilot-scale trials will be needed to confirm the predicted treatment performance and refine the estimated volume for the full-scale system, including pond volume and volumes for each stage of the bioreactor. The benefits of supplying liquid organic carbon should be evaluated and reagent dosage should be determined for full-scale operation. Finally, the composition of the matrix will need to be refined, based on information in the published literature and on laboratory and/or field tests. This information will allow the company to determine the footprint of the full-scale system, determine capital and operating costs, and establish the longevity of the organic matrix and plan for its operation and maintenance.

## NORTH WETLAND DESIGN - PREDICTED PIT LAKE DRAINAGE CHEMISTRY

Source Environmental Associates (SEA) conducted the modeling that predicts chemical composition of the pit lake discharge at closure. The pit lake is estimated to take approximately one hundred years to fill to the point of overflow. At that time, pit lake water overflow will be directed into the North TMF Wetland for treatment, as shown in Figure 1.

Knight Piésold Ltd. (KP) predicted flow rates for the mine discharge, based on meteorological and hydrological characteristics of the site. Discharge from the pit lake will be controlled, and will only be released during the biologically active summer months of June through September. The rate of release will be approximately 180 l/s (KP, 2013). To ensure sufficient wetland design, the inflow rate was increased by 20% to account for any uncertainty in the pit lake discharge rate. The resultant design flow rate of 220 l/s (0.22 m<sup>3</sup>/s) will be used in the calculation of wetland area.

SEA also predicted the average inflow water quality to the North TMF Wetland. Table 6 shows that concentrations of cadmium, copper, molybdenum and uranium are elevated relative to the CCME Guidelines for Protection of Freshwater Aquatic Life. The cadmium and copper guideline values were adjusted for hardness using the formula to calculate hardness-adjusted Guidelines provided by CCME<sup>10</sup>. Comparison of predicted pit lake discharge concentrations and CCME guidelines show that cadmium exceeds its guideline value by a factor of 50; copper by a factor of 93; while molybdenum and uranium exceed CCME by factors of less than 5. Maximum predicted inflow concentrations for these constituents will be used for a conservative calculation of the area required for a treatment wetland.

Table 6 also shows that selenium concentrations are predicted to be 0.0085 mg/L, below the 0.010 mg/L levels considered to be the threshold for risk of ecotoxic effects to shorebirds and waterfowl in wetlands, as discussed further in the South TMF Wetland section. In addition, mercury concentrations are predicted to be 0.000025 mg/L, below CCME Guideline levels.

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<sup>10</sup> These equations are: Cadmium concentration =  $10^{0.86[\log_{10}(\text{hardness})]-3.2}$  µg/L and Copper concentration =  $e^{0.8545[\ln(\text{hardness})]-1.465}$  \* 0.2 µg/L, taken directly from the CCME web site [<http://st-ts.ccme.ca/>]

**Table 5. Predicted Average Inflow Water Quality to the North Wetland<sup>1</sup>**

Parameter	Pit Lake Discharge	CCME Water Quality Guideline
Sulphate (SO <sub>4</sub> )	474	-
Hardness (Total)	468	-
Aluminum	0.0087	0.10
Arsenic	0.005	0.005
Cadmium (Cd)	0.0039	0.00008 <sup>3</sup>
Copper (Cu)	0.37	0.004 <sup>3</sup>
Mercury (Hg) <sup>2</sup>	0.000023	0.000026
Molybdenum (Mo)	0.18	0.073
Selenium (Se) <sup>2</sup>	0.0083	0.001
Silver (Ag)	0.0007	0.001
Uranium (U)	0.062	0.015

<sup>1</sup> Taken from Source Environmental spreadsheet: Results\_Dec\_11\_SEA\_Casino\_WQM\_v2 emailed December 11, 2013.

<sup>2</sup> Values for mercury and selenium account for treatment by the bioreactor during heap draindown.

<sup>3</sup> Value corrected for hardness.

## FEED CHEMISTRY AND DESIGN CRITERIA FOR PIT LAKE DRAINAGE

The above discussion defines the chemistry and design criteria for a pit lake drainage treatment system at the Casino Project. The wetland will be designed to treat cadmium, copper, molybdenum and uranium, all of which have concentrations predicted to exceed the CCME Guidelines, as indicated in Table 7. Treatment of sulphate by the wetland will also be considered.

*Table 6. Worst case predicted pit lake drainage concentrations for cadmium, copper, molybdenum and uranium, and their acceptable concentrations in freshwater.*

Constituent	Worst Case Concentration	CCME Guidelines <sup>1</sup>
<b>Cadmium</b>	0.0039 mg/L	0.00008 mg/L
<b>Copper</b>	0.37 mg/L	0.004 mg/L
<b>Molybdenum</b>	0.18 mg/L	0.073 mg/L
<b>Sulphate</b>	477 mg/L	-
<b>Uranium</b>	0.062 mg/L	0.015 mg/L

<sup>2</sup> Guidelines for cadmium and copper corrected for hardness

The pit lake overflow will discharge into the North TMF Wetland from June to September. The design flow rate used to size the North TMF Wetland is 0.22 m<sup>3</sup>/sec, which includes a 20% safety factor increase over modelled flow.

## PROCESS CHEMISTRY

### *Metal Removal*

Wetlands effect metal removal by a variety of physical, chemical, and biological processes. Plants play an essential role in forming and maintaining wetland sediments, but they do not accumulate metals and are generally recognized to account for less than 5% of their removal. The most significant mechanisms involved in metal removal include:

- Sorption and/or exchange onto organic matter [detritus];
- Formation of carbonates;
- Association with iron and manganese oxides;
- Reduction to non-mobile forms;
- Formation of insoluble metal sulphides;

The affinity of metals for organic matter and their sorption onto it are well-documented (Kadlec and Keoleian, 1986). Given the abundance of organic matter in the detritus layer of wetlands, it clearly plays an important role in metal retention. Three metals in particular are known to have an especially strong affinity to organic matter and to be retained in wetlands: copper, uranium and nickel.

“Cupriferous” and “Uraniferous” bogs have been reported by exploration geologists for many years. One of the most striking reports concern the copper retained in Tantramar Swamp, in Sackville, New Brunswick, Canada (Boyle, 1977). Copper concentrations in the black muck of this one hectare wetland reach an astounding 5-9%. The bulk of this copper is associated with organic matter, on account its resistance to most extractants. Interestingly, this wetland contains approximately 300 tons of copper and is estimated to have been accumulated it ever since it was formed 4,000 years ago, based on existing metal loading rates (Fraser 1961).

Several “Uraniferous bogs” have been documented in the Western States in the 1980’s by the United States Geological Survey (Owen and Otton, 1995). One example is the uraniferous wetlands in the north fork of Flodelle Creek, in Northeastern Washington, which accumulate uranium in their organic-rich sediments. Laboratory studies indicate that the uranium is strongly bound to organic matter in the detritus layer and is biologically unavailable (Zielinski and Meier, 1988)<sup>11</sup>.

In studies of other natural wetlands, copper has been shown to accumulate as copper sulphides as well in an organically-bound form. The presence of sulphate in water and the existence of strongly-reducing sediments appear to govern its partitioning into one or the other phase. In a study conducted at the Bell Copper Mine, it was shown that copper is initially retained through interactions with organic matter, but that it eventually converts to

<sup>11</sup> A similar finding was made for uranium retained in sediments of Island Lake fen, which effectively treated the discharge from the Cluff Lake Mine (See below).

insoluble sulphides (Sobolewski, 1996b). This finding is significant because if copper is predominantly retained as sulphide minerals, then the duration of treatment will be indefinite as long as wetland sediments are kept reduced.

A similar finding was reported by the Canadian geologist Robert Boyle (Boyle, 1965), who noted the following while exploring the Keno-Hill area:

“Streams and springs that dissipate their water into bogs have their zinc (as well as other metals) largely removed. Initially this zinc is loosely bound (but) with aging, the zinc partakes of the organic colloidal complexes and is then (...) unavailable to most extractants.”

Another study by Sobolewski (2006) demonstrated that uranium retained in the organic sediments of the Island Lake fen, below the Cluff Lake mill, resisted extraction by acid leach or even release after sediment desiccation. Molybdenum was similarly strongly retained in these sediments, presumably as molybdenite ( $\text{MoS}_2$ ), since it was only released on incubation with a 35% peroxide leach solution.

These different observations describe an aging phenomenon that has not been fully documented. The organic matter that retains metals may release them upon decomposition, allowing for the formation of sulphides (Berner, 1980). Alternatively, progressive decomposition may expose new functional groups which form additional, stronger bonds with metals. Whatever the process, this aging phenomenon is extremely important, as it pertains to the long-term treatment performance of wetlands and the fate of retained contaminants. In addition, greater resistance to extractants indicates that contaminants become biologically unavailable, as indicated above.

Cadmium and zinc also interact with clays and oxides of iron and manganese. Their 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.

The retention of metals on iron and manganese oxides is potentially reversible because these oxides can resolubilize under anoxic conditions. Under these conditions, the metals would be released back to surface waters. This can be a problem in treatment wetland, where oxides that are formed at the wetland surface can redissolve during the winter, when the ice cover creates anoxic conditions. This scenario will not develop at Casino because iron and manganese concentrations entering the North Wetland are very low (Table 6).

Another important process for metal removal in wetlands is the formation of highly-insoluble sulphides (Table 4). It is evident that the sulphides of copper, cadmium and molybdenum have exceedingly low solubilities, which means that they can be removed to very low concentrations.

This provides the basis for biologically-based treatment systems that use sulphate-reducing bacteria (SRB). These bacteria use sulphate instead of oxygen to oxidize organic matter, producing bicarbonate and hydrogen sulphide as metabolic by-products. The latter reacts



with metals to render them insoluble. There have been numerous reports indicating that this is a key process that acts in wetlands to remove toxic metals (Machener and Wildeman, 1992; Sobolewski, 1996b; Gammon et. al., 2000).

Uranium geochemistry was discussed earlier in the context of the treatment of heap draindown. In the reducing environment of wetland sediments, uranium is rapidly and strongly retained on decaying organic matter (detritus) as well as the reduced, immobile uraninite.

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, copper or molybdenum 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. This is a key role played by the detritus layer in a treatment wetland.

In summary, cadmium, copper and molybdenum are largely removed from solution in wetlands by two processes: adsorption onto organic matter and formation of insoluble sulphides. Uranium is also removed as the insoluble uraninite. Some studies have shown that organically-bound metals will age into forms highly resistant to extraction and are biologically-unavailable. Metals that are retained with iron and manganese oxides are more labile, but this process is not expected to occur in the North TMF Wetland.

#### *Treatment Wetlands*

Wetlands are stagnant, transitional, highly-productive ecosystems that develop in water-logged or flooded, gently-sloping lands. Their capacity to remove metals from mine drainage is well documented (Sobolewski, 1999). 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 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 and organic carbon 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 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<sup>2</sup>/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<sup>3</sup>/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 source of organic material, regional climate, accessibility for maintenance, etc. All these factors will be relevant and important for the Casino Project.

#### *Case Studies – Natural Wetlands*

There are many examples of natural wetlands removing cadmium, copper, molybdenum, uranium and zinc from mine drainage, including wetlands below the Con Mine (Ball, 1993), a wetland downstream of an abandoned lead/zinc mine in Glendalough, County Wicklow, Ireland (Beining and Otte, 1996, reviewed in Sobolewski, 1999), the Halamanning wetland in the St Hilary Mining District (Cornwall, UK; Brown 1997), a wetland below the Pacific Mine, Utah, USA (Lidstone & Anderson, 1993, reviewed in Sobolewski, 1999), numerous wetlands in the Keno Hill Mining District (Sobolewski, 1999), and wetlands below the Tom's Gully Mine and below the Woodcutter Mine, Darwin, Australia (Noller et al., 1994). In fact, a survey of over 60 natural wetlands documented to provide partial or complete

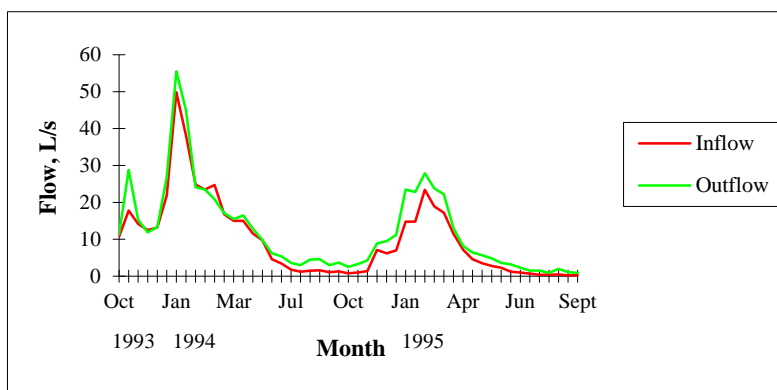
treatment of contaminated mine drainage, copper was removed in every wetland that was examined. Uranium, where present, was also always removed.

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.

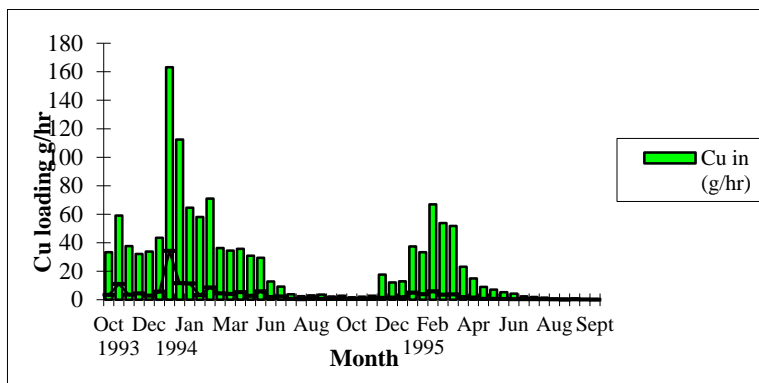
A number of case studies are reviewed below to provide more detail on their characteristics and performance. The emphasis of these reviews is to extract useful information and quantitative data that can be used to design the North Wetland or predict its performance.

### **Halamanning Wetland, Cornwall, UK**

The 2.7 ha Halamanning natural wetland (Cornwall, U.K.) has been steadily removing copper from mine water flowing through abandoned adits in the St Hilary Mining District. Copper removal is consistently >80%, on flows varying seasonally from >1.0 to 50 L/sec (See Figure 7 and Figure 8, taken from Brown, 1997). Copper removal was strongly related to loading rates in this wetland. Loadings of 1.4 kg Cu/ha/d gave 80% removal, whereas loadings of 0.57-0.96 kg Cu/ha/d gave >90% removal. In the latter case, the wetland decreased copper concentrations from 0.80 mg/L to 0.070 mg/L, on a retention time of 3.7 days. Note also this removal occurred during the winter.



*Figure 7. Inflows and outflows for the Halamanning wetland.*



*Figure 8. Copper concentrations in inflows and outflows of the Halamanning wetland.*

Copper in this wetland appears to be retained both by sorbing onto organic matter and by forming insoluble sulphides.

The above data indicate that copper removal averaged 0.471 kg/ha/d. This is a year-round average removal rate that is dominated by periods of high flow during the rainy half of the year.

### **Star lake Mine**

The operating Star Lake gold mine (Northern Saskatchewan) discharged its tailings pond supernatant onto muskeg during the ice-free season to remove copper (Gormely et. al., 1990). This supernatant was spiggotted separately onto two fens measuring 0.74 and 25 hectares, respectively. Input copper concentrations in 1990 ranged from 0.8 to 2.2 mg/L, on mean flows of 1,305 m<sup>3</sup>/day (15 L/sec). Copper removal rates exceeded 98% on average mass loadings of approximately 1 kg/ha/day, which yield an areal removal rate of more than 0.98 kg/ha/day. Such a removal rate is comparable with that calculated for the Halamanning wetland described above.

### **Equity Silver Mine**

The operating Equity Silver Mine discharged water containing residual copper concentrations into Bessemer Creek. This water flowed through a natural wetland (Buck Creek swamp) that covered approximately 0.3 hectares. In 1984, copper loads to the wetland ranged from 1.9 to 5.5 kg/ha/day, and removal rates of 64-79% were recorded, with discharge concentrations of 0.003-0.0046 mg/L. This yields average areal removal rates of 2.1 kg/ha/day.

### **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 (Noller *et al.*, 1994). The circumneutral discharge from the mine flows at somewhat less than 120 L/min and contains several metals (Table 8). 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<sup>12</sup> (P. Woods, ERA Ranger Mine, Jabiru, NT, Australia. Personal communication).

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<sup>12</sup> 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).

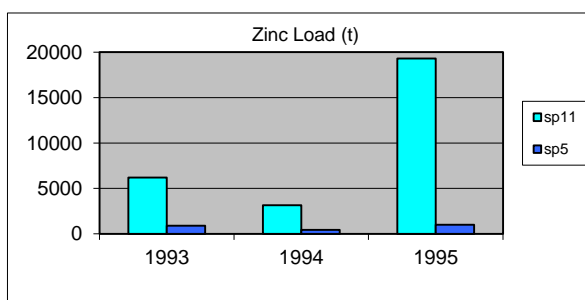


Figure 9. Zinc loads before and after natural wetland at Woodcutter's Mine.

Assuming an effective area of 10 hectares (100 x 1,000 m) for the lower half of the wetland, and assuming flows of 110 L/sec, this yields areal removal rates for cadmium of 0.05 kg/ha/day and areal removal rates for zinc of 4.9 kg/ha/day. Note that the removal rate for cadmium appears to be minimal because the system is loading-limited for this metal. That is, higher removal rates would have been measured if cadmium concentrations and/or loading had been higher.

Table 7. 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 area of 1.5 hectare, this wetland was removing zinc at a rate of 0.09 kg/ha/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

Several natural wetlands in this historic mining district have been shown to remove metals from various adits discharging at the site.

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 9). In others, zinc was retained as a sulphide.

*Table 3. Metal concentrations in wetland sediments and plants in the Keno Hill area<sup>a</sup>.*

	S. McQuesten* swamp	Galkeno natural swamp	Non-impacted <sup>b</sup> sites	Non-impacted <sup>b</sup> sites
Metal	Sediments/Plants <i>n</i> =2	Sediments/Plants <i>n</i> =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

<sup>a</sup>Data expressed as mg/dry kg

<sup>b</sup> Ranges and means of concentrations of metals in aquatic grasses and forbs and sediments from non-impacted wetlands, as reported by Hutchinson, 1975

Table 8. Concentrations of selected metal species in sediments of the Galkeno swamp<sup>a</sup>.

	Wash	Organic	Carbonate	Fe + Mn	Sulphides	Residue	Total
Cd	<0.5 <sup>b</sup>	<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

<sup>a</sup> Taken from Sobolewski, 1996c.

<sup>b</sup> Data are expressed as mg/dry kg

In a study conducted in October 2001, zinc concentrations, water pH and temperature were 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 °C. 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 °C to 0.2 °C 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 10). 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 10).

Table 9. 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

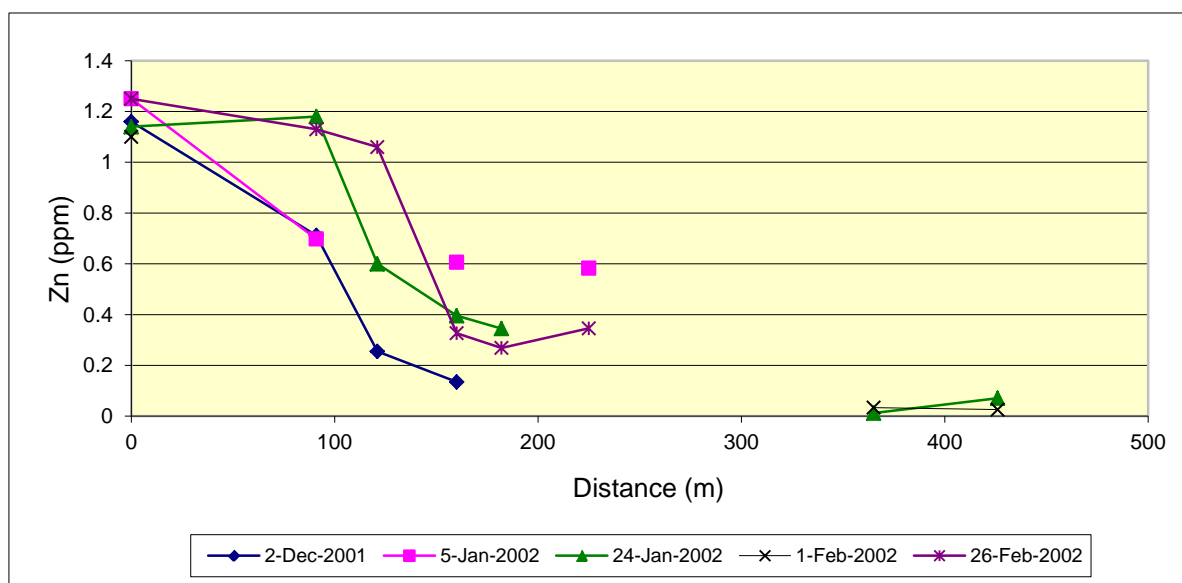


Figure 10. Zinc concentrations below Silver King adit, Elsa Camp.

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<sup>2</sup>/d. Although this is much higher than the areal removal rates in the other examples, it is important to note that some of this removal was due to co-precipitation with iron oxides and oxyhydroxides (which were evident during sampling).

### Island Lake Fen

The discharge from the Cluff Lake uranium mine and mill (Northern Saskatchewan) flowed through the Island Creek drainage, which includes a number of small lakes and a small fen along its flow path. This ~5 hectare fen has a typical flora of Northern Boreal climate, with hummocks dominated by cattails (*Typha latifolia*) and water sedges (*Carex aquatilis*) (TAEM 1994). It is located quite a distance downstream from the mill (a few kilometers) and received flows from both the mill effluent and the natural drainage in the area<sup>13</sup>. Flows through this natural fen averaged 67.3 L/sec. Its ability to remove molybdenum and uranium was notable. CNSC reported that: "the filtering effect of the Island Lake Fen reduced molybdenum concentrations by 99% in water downstream at Island Creek (reduced to 0.014 mg/L from a concentration of 1.0 mg/L at the Island Lake outlet)" (CNSC, 2003). This provides a removal rate of 1.14 kg Mo/ha/day. Uranium was also removed by 98-99%, from inflow concentrations of 0.15-0.30 mg/L to outflow concentrations averaging 0.0015 mg/L, which yields a removal rate of 0.23 kg U/ha/day. Note that these rates are likely minimum removal rates because the concentrations at the wetland discharge were very low and the system appears to have been load-limited.

<sup>13</sup> During operation, mill effluents varied from 350,000 – 1,147,226 m<sup>3</sup>, which is estimated to comprise 33% of the total drainage flowing through the fen.



Follow-up studies demonstrated that both metals were stably retained in the wetland sediments as molybdenum sulphides and immobile uraninite, respectively (Sobolewski, 2006). Additional studies indicated that neither molybdenum nor uranium were taken by plants and transferred through the food chain (Canada North, 2004).

#### *Case Studies – Constructed Wetlands*

The 9.3 hectare treatment wetland at the operating Campbell Mine, located in the Red Lake Mining District, Northern Ontario, receives water from a polishing pond during the ice-free season and discharges into nearby Balmer Lake. This wetland has operated since 1999 and has an excellent record of inflow/outflow water chemistry because it is the legal discharge point for the mine.

The wetland influent contains copper concentrations ranging from 0.01 to 0.05 mg/L and discharges at concentrations ranging from 0.002 to 0.008 mg/L. Flows into the wetland are very large, reaching up to 20,000 m<sup>3</sup>/day. Areal removal rates for copper have been calculated at 1.5-3.5 kg/ha/day.

Several small-scale wetlands were constructed at the Dunka (Taconite) Mine in Minnesota to remove nickel on a year-round basis, although they also removed cobalt, copper, and zinc (Eger *et. al.*, 1991). The wetlands were surface flow wetlands planted with cattails and sedges, ranging in surface area from 0.25 to 1.7 hectares. They received flows from 0.33-5.2 L/sec. Inflow and outflow concentrations for these metals differed considerably for each of these wetlands.

Table 11 shows data for copper at the five wetlands on the site. The wetlands at W2D-3D and at W1D removed copper very effectively, whereas the other wetlands were less effective in removing copper. Unfortunately, the data from these wetlands do not allow calculating areal removal rates because copper concentrations at the discharge were at or below detection limits. The other wetlands were much less effective in removing copper, which reflected design problems.

*Table 10. Influent and effluent copper levels for five wetlands at the Dunka LTV Mine.*

Wetland site	Influent (mg/L)	Effluent (mg/L)
W2D-3D	0.06	<0.005
W1D	0.02	<0.005
Seep 1	0.27	0.06
Seep X	0.41	0.14
EM-8	0.03	0.01

W1D received drainage also 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 0.4 kg Zn/ha/d. However, this rate is expected to be a minimum because the system was loading-limited.



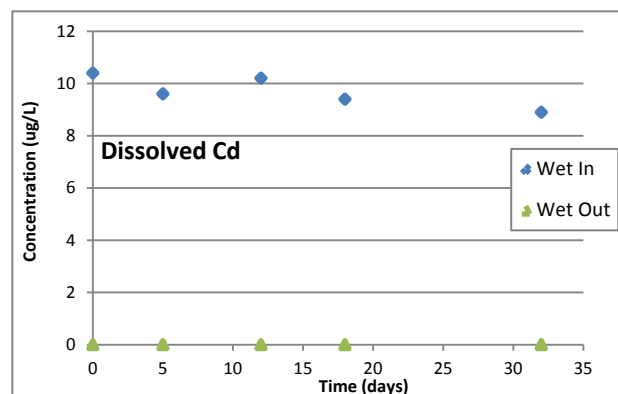
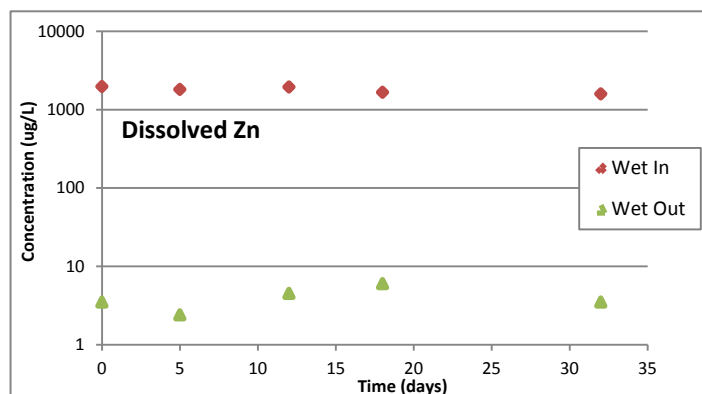
There are few reports of constructed wetlands that remove cadmium and zinc which provide the information needed to calculate removal rates. Additionally, few investigations measure cadmium and zinc concentrations using low or sub-ppb detection limits. Typically, cadmium is measured with detection limits of 0.5 ppb, which is barely sufficient for the Casino 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 donor 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 kg/ha/day.

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 Casino 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<sup>14</sup>. 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 11). 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. Copper was present at 0.030-0.050 mg/L in this system and was removed to below detection limits (which prevents calculating removal rates).

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.



<sup>14</sup> Sobolewski, A. Personal observations.

*Figure 11. Influent and effluent Zinc and Cadmium concentrations in pilot-scale treatment wetland.*

The volumetric removal rates for these metals are  $0.00232 \text{ g/m}^3/\text{d}$  for cadmium and  $0.434 \text{ g/m}^3/\text{d}$  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.

The ERA Ranger mine has experimented with treatment wetlands since 1987. Pilot studies conducted from 1990 to 1994 indicated that up to 90% of input uranium could be retained in treatment wetlands, while other dissolved salts or metals were either unaffected or actually increased in concentration. An inordinately abundant rainfall in the 1994/95 monsoonal wet season nearly exhausted the mine's water storage capacity. This led to a decision to construct a full-scale treatment wetland.

A series of nine wetland cells covering six hectares were constructed in 1995. Wetland sizing was based on the empirically-derived removal rates obtained in the pilot wetlands. The wetland cells were planted with tubers of *Nymphae* spp. and rhizomes of spike rush (*Eleocharis sphacelata*), with the latter eventually becoming dominant. The cells were originally designed to be separated with permeable berms made of waste rock from the mine site. However, this material released considerable amounts of salts and manganese from weathering, and sandy/clay was used ultimately.

Uranium in leachate from ore stockpiles reported at  $400\text{--}1,200 \text{ }\mu\text{g/L}$ . During the five-month trial started in July 1995,  $450,000 \text{ m}^3$  of runoff was circulated through the system (average flow rate of  $40\text{--}60 \text{ L/sec}$ , pH 7.5). Its outlet concentrations decreased by over 80% for the first month of the trial, and by 40%–60% thereafter. Uranium removal rates improved in subsequent years and the constructed wetland presently removes uranium very effectively (SSG, 1998).

Water from another retention pond was discharged into the Djalkmara billabong<sup>15</sup>, a natural wetland on the property. It was hoped that the constructed wetland could be made to function similarly to this wetland, since it had removed uranium effectively for the past ten years. For instance, it was known that sediments from the Djalkmara billabong quickly remove uranium when added at concentrations found in mine water (Figure 12).

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<sup>15</sup> A wetland flooded during the wet season, subsequently releasing water until it dries out.

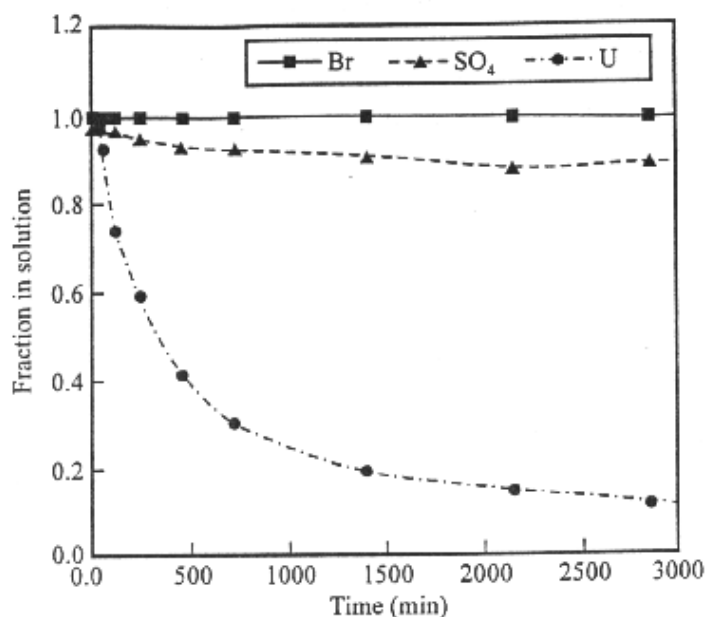


Figure 12. Changes in bromide, sulphate, and uranium in water overlying sediments from the Djalkmara Billabong (Taken from Jones et al., 1996).

The rapid removal of uranium from the water column by wetland sediments (e.g., Figure 12) suggests that it is sorbed onto organic matter and/or iron oxides. Laboratory studies have suggested that this uranium might be re-mobilized if exposed to high bicarbonate concentrations (Zielinski and Meier, 1988; Owen and Otton, 1995). Bicarbonate concentrations do increase during the dry season in the Djalkmara billabong, but there is no concomitant solubilization of uranium from its sediments (Noller et al., 1997), suggesting that this phenomenon does not occur at environmentally relevant bicarbonate concentrations. Thus, the wetland can be considered to be an effective filter for uranium dissolved in mine water.

Although there are no design criteria for wetlands treating uranium, its rapid removal and similar geochemical behaviour to copper suggests that a wetland sized for copper removal will also effectively remove uranium, so long as similar removal rates are required.

#### *Sizing criteria for a treatment wetland for Casino*

The above reviews described both natural and constructed wetlands that remove cadmium, copper and/or zinc from mine drainage. Data from several of them could be used to derive areal removal rates, though some caveats are necessary in applying these rates where the wetlands are loading-limited (i.e., they received smaller metal loadings than they are able to treat effectively).

These data are summarized below and assessed for their applicability in sizing the North Wetland. The assessment for copper will be separated from that for cadmium and zinc.

## Cadmium

The areal removal rates for cadmium and zinc for various wetlands described above are tabulated below (Table 12).

*Table 11. Cadmium and zinc removal rates for various natural and constructed wetlands.*

Site	Ave input Cd (mg/L)	Areal removal rate (kg Cd/ha/d)	Ave input Zn (mg/L)	Areal removal rate (kg Zn/ha/d)
Woodcutter's Mine	0.063	0.05	6.9	4.9
Silver Queen Mine	NA	NA	4.04	0.09 (winter)
Silver King Mine	NA	NA	1.2	77.8 (winter)
Natural wetland (Yukon)	NA	NA	3.2	0.59
Constructed Wetland (Minnesota)	NA	NA	0.52	0.4
Constructed Wetland (Montana) <sup>1</sup>	0.0327	0.0031	9.33	0.898
Constructed Wetland (Yukon)	0.0066	0.018	25	31
Constructed Wetland (Colorado) <sup>1</sup>	0.0099	0.0023	1.85	0.434

<sup>1</sup>Volumetric rates (g/m<sup>3</sup>/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 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. This rate will be excluded from consideration, since iron oxides will not make a significant contribution in the North Wetland.

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. **Taking a median value, rather than an average rate, a conservative areal removal rate of 2 kg Zn/ha/d will be used in calculating the surface area requirements for zinc removal by the North Wetland.**

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.

**Since there is uncertainty in the rate for cadmium removal, it is set to 1/20 that for zinc, giving a conservative areal removal rate for cadmium of 0.1 kg Cd/ha/d.**

## Copper

The areal removal rates for copper for various wetlands described above are tabulated below (Table 13).

These removal rates are remarkably consistent, despite the diversity of wetlands that treat mine waters. The low removal rate at the Halamanning wetland likely reflects the inefficiency of this natural system. The removal rates for the other wetlands span a narrow range of 1-3.5 kg/ha/day at temperatures comparable to those at Casino (6-10 °C).

**Conservatively, a removal rate of 1 kg/ha/day will be used in calculating the surface area requirements for copper removal by the North Wetland.**

*Table 12. Copper removal rates for various natural and constructed wetlands.*

Site	Ave input Cu (mg/L)	Areal removal rate (kg Cu/ha/d)
Halamanning wetland	0.80	0.471
Star Lake Mine	0.8-2.2	0.98
Equity Silver Mine Wetland	0.019	2.1
Campbell Mine Wetland	0.01-0.05	1.5-3.5

## Molybdenum

There is a paucity of information that allows a calculation of areal removal rates for molybdenum. Data from the Island Lake fen indicated that it was removed at a nominal rate of 1.14 kg/ha/day. This is presumed to be a minimum removal rate because there was likely short-circuiting in this natural wetland. Nonetheless, given the similar climactic conditions between this fen and the area for the Casino Project, this rate is judged to be acceptable for design purposes.

## Sulphate

For completeness, the removal of sulphate is discussed here. It has been observed repeatedly that some sulphate is removed by wetlands treating mine drainage by 10-30%. The factors that govern its retention in wetlands have not been investigated. Clearly, some sulphate will likely be retained as insoluble metal sulphides. Some sulphate will also be retained as elemental sulphur, deposited at the oxic/anoxic sediment interface. A value of 15% was recommended as a reasonable removal rate for the North Wetland, based on professional judgement.

## Uranium

Data from the Ranger Mine were not used to calculate uranium removal rates because their wetlands are not comparable in either operation (seasonal operation during wet months) or composition (tropical vegetation). However, data from the Island Lake fen can be used and a removal rate of 0.23 kg U/ha/day was obtained. As for molybdenum, this rate will be used for design purposes.

The above removal rates assume that metal removal will occur by a combination of adsorption onto organic matter and reaction with biogenic hydrogen sulphide generated in wetland sediments. The latter assumption is predicated on the presence of adequate sulphate concentrations in mine drainage. Table 6 indicates that sulphate concentrations are predicted to range from 309 to 358 mg/L in the inflow to the North Wetland. This is more than the minimum concentration of 250 mg/L  $\text{SO}_4$  required to sustain full bacterial sulphate reduction. Another requirement of bacterial sulphate reduction is that water pH should be more than 5.0 and less than 9.5. While the Source Environmental water quality modeling did not predict a specific pH for water flowing into the North Wetland, it did indicate that it will remain above pH 5 after the pit lake overflows, reflecting the excess alkalinity predicted in this water. Thus, there are no parameters predicted in the North Wetland inflow that would preclude the use of a treatment wetland.

There are no plans to operate the North Wetland during the winter, so winter removal rates are not needed to calculate its surface area. However, the above data indicate that metal removal still occurs during the winter at a rate approximately one-tenth that of summer time. This provides a measure of contingency, should there still be water flowing through the wetland during the winter months.

## CASINO NORTH TMF WETLAND TREATMENT SYSTEM

The size of the North TMF Wetland is determined from the areal removal rates established from the above review of case studies. Its surface area is determined by calculating flows and contaminant loadings entering the wetland and dividing them by areal removal rates. This calculation is presented below.

**The design flow rate for water flowing into the North TMF Wetland has been set at 0.22 m<sup>3</sup>/sec, based on contributions from the pit lake and runoff from surrounding areas.**

Worst case scenario metal concentrations for water flowing in the North TMF Wetland are presented in Table 7. Guideline concentrations are subtracted from input concentrations, and the resulting amount of metal to be removed is multiplied by the design flows to calculate metal loads to be removed. These metal loads are presented in Table 14.

*Table 13. Metal loads to be removed from inflows to the North Wetland.*

Contaminant	Daily load to be removed
Cadmium	72.6 g/day
Copper	6.96 kg/day
Molybdenum	2.03 kg/day
Uranium	0.89 kg/day

Design removal rates were calculated above and are presented in Table 15 for each contaminant of concern. These rates are divided by the daily metal loads that must be removed (Table 14) to determine the wetland surface area required for treatment, which are also presented in Table 15.

*Table 14. Design removal rates and surface area required for the North Wetland.*

Contaminant	Areal Removal Rate	Surface area required
Cadmium	100 g/ha/day	0.73 hectare
Copper	1 kg/ha/day	7.0 hectare
Molybdenum	1.14 kg/ha/day	1.8 hectare
Uranium	0.23 kg/ha/day	3.9 hectare

The above calculations show that copper and molybdenum have the highest loadings of all four metals. The wetland area required to remove copper is 7.0 hectares, an area that would also remove uranium, cadmium and molybdenum. Therefore, a wetland designed to remove copper will also be capable of fully removing cadmium, molybdenum and uranium from mine drainage. This surface area is calculated on the basis of worst-case metal concentrations and maximum flows, and is therefore highly conservative.



Sulphate is predicted to be retained by a factor of 15% in the North TMF Wetland. This will decrease its concentrations from 474 to 403 mg/L.

The wetland area planned over the TMF is shown in Figure 1. This area may be revised in the future, based on updated site information, such as predicted drainage chemistry or site-specific removal rates. It is recommended that an on-site pilot study be operated over several years during the active period of the mine life to prove out the predicted removal rates.

The previous studies at Keno Hill, Island Lake fen and elsewhere indicate that metals removed in treatment wetlands are retained in their sediments and are not taken up in plant tissues. This is an important consideration because a wetland will be a new element in the post-closure landscape and will likely attract new wildlife populations. The absence of metal uptake in plant tissues means that metals retained in the wetland will not enter the food chain and affect these new populations.

The expected presence of resident populations (such as beavers and muskrats) and the attraction of marsh birds and moose will have to be accounted for in the wetland design. Thus, the wetland design will use gentle slopes that are less susceptible to destruction by resident mammals. The wetland edges will be curved and follow the contour of the land, reflecting its ecological function as a littoral zone to the lake created by the TMF. It will be planted with a mix of vegetation with preferences for shallow and deep water depths. However, large areas of open water or exposed shoreline are undesirable, as they are more prone to reworking sediments and potentially releasing metals.

The above wetland features are not meant to supersede the primary goal of metal removal. Instead, they are intended to support the long-term integrity of this new landscape unit by providing a design that promotes and maintains a healthy ecosystem, while remaining compatible with a design for metal removal.



## CASINO SOUTH TMF WETLAND TREATMENT SYSTEM

Groundwater is predicted to upwell through the submerged waste rock within the TMF and discharge at its surface. The concentrations of several metals in this drainage are elevated, notably cadmium, copper, mercury, molybdenum and uranium. In addition, selenium exceeds existing guidelines. Most of these contaminants only exceed Guidelines by a factor of less than 10, but copper exceeds its guideline by a factor of 25. These excessive metal concentrations will be decreased by conveying this water through a treatment wetland prior to discharge from the TMF pond, much as has been proposed for treatment of the pit lake discharge. For this reason, the background information given for the design of the North TMF Wetland will not be repeated here.

### FEED CHEMISTRY FOR SOUTH TMF WETLAND

The above information and discussion on the design criteria for the North TMF Wetland is entirely applicable to the design of the South TMF Wetland. The South TMF Wetland will be designed to treat cadmium, copper, mercury, molybdenum and uranium concentrations as they are predicted to exceed the CCME Guidelines, as indicated in Table 16.

**Table 15. Predicted Inflow Water Quality to the South TMF Wetland<sup>1</sup>**

Parameter	TMF Pond (95 <sup>th</sup> percentile)	CCME Water Quality Guideline
Sulphate (SO <sub>4</sub> )	482	-
Hardness (Total)	554	-
Cadmium (Cd)	0.00054	0.00008 <sup>2</sup>
Copper (Cu)	0.133	0.004 <sup>2</sup>
Mercury (Hg)	0.000016	0.000026
Molybdenum (Mo)	0.136	0.073
Selenium (Se)	0.0055 (average 0.0045)	0.001
Silver (Ag)	0.00006	0.001
Uranium (U)	0.052	0.015

<sup>1</sup> Taken from Source Environmental: email dated December 11, 2013.

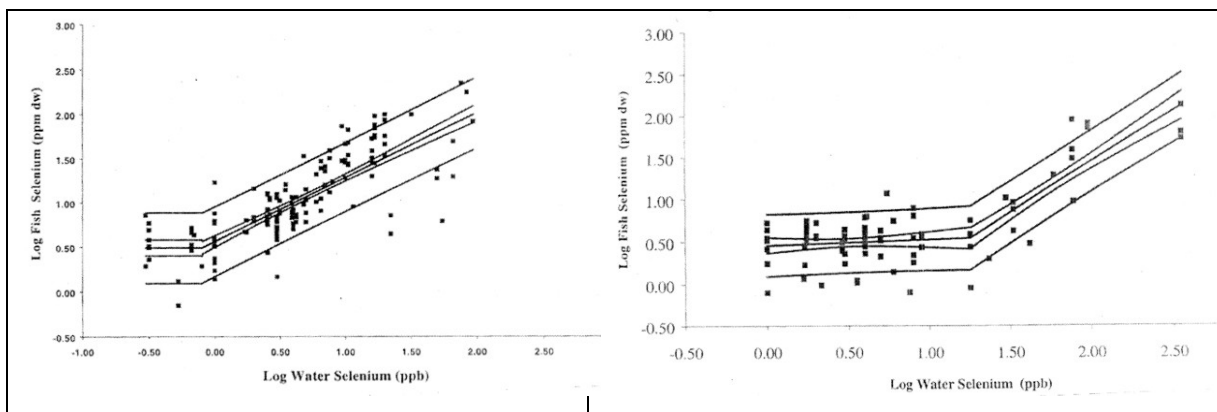
<sup>2</sup> Value corrected for hardness.

Selenium concentrations also marginally exceed the Guidelines, but this contaminant is a special case. Existing BC and CCME guidelines for selenium are known to be inappropriate, since its ecotoxicity depend on the physiographic and ecological characteristics of the receiving environment more than on its ambient concentrations (SETAC, 2010). Given that selenium is known to be removed by wetlands, but that it can potentially enter the food chain (Fan and Higashi, 1997), the idea of using a wetland to treat the TMF drainage must be evaluated carefully.

Two important factors govern the behaviour and ecotoxicity of selenium in aquatic environments.

1. Selenium exists in surface waters in either of two species: selenite ( $\text{SeO}_3^{-2}$ ) and selenate ( $\text{SeO}_4^{-2}$ ). While both oxyanions can exert toxic effects, selenite is fairly reactive and will readily be removed from the water column. In contrast, selenate is unreactive and more mobile. Thus, the selenium species governs the extent of its travel from a source and area of potential effect.
2. In general terms, lakes, reservoirs and other quiescent water bodies (“lentic ecosystems”) favour selenium accumulation in the food chain, whereas fast-flowing water bodies (“lotic ecosystems”) do not.

The difference between lotic and lentic ecosystems has been observed repeatedly. An earlier review noted that selenium accumulates at much lower concentrations in fish sampled in lentic environments compared with lotic environments (See Figure 13, taken from Adams et. al., 2000).



*Figure 13. Regressions of water selenium vs whole fish selenium concentrations. Left: lentic environments; right: lotic environments.*

In the above comparison, selenium begins to accumulate in fish tissues from lentic environments when its concentrations exceed 0.001 mg/L. In contrast, selenium begins accumulating in fish tissues from lotic environments when its concentrations exceed 0.018 mg/L. In addition, the slope of these regressions is steeper for lentic compared with lotic environments. An increase of 0.001 to 0.0033 mg/L in lentic environments causes an approximate doubling of selenium concentrations in whole fish, whereas the same doubling is only achieved when its concentrations increase from 0.018 to 0.033 mg/L in lotic environments. It must be acknowledged that shorebirds and waterfowl, rather than fish, are the primary target for ecotoxic effects in wetlands. This information suggests that wetlands may be susceptible to ecotoxic effects from selenium, but the range of concentrations that show effects on shorebirds and waterfowl must be evaluated from the literature. In addition, this information predicts that 0.010 mg/L selenium will have no effects in Casino Creek (a lotic environment).

The TMF outflow to the South TMF Wetland contains selenium, averaging 0.0045 mg/L, with a 95<sup>th</sup> percentile value of 0.0055 mg/L. These levels are below the concentrations

shown to have an effect on shorebirds and waterfowl in the Richmond refinery treatment wetland, which ranged from 0.010 to 0.30 mg/L. At that site, water flowed through wetland cells covering an area of 36 hectares. Selenium uptake, but not ecotoxic effects, was reported for red-wing blackbirds in the Goddard Marsh and Clode Pond (Elk River Valley, BC), which received 0.012-0.018 and 0.071-0.093 mg/L, respectively (Chapman et. al., 2008). Uptake was also noted in amphibians collected in these ecosystems, but evaluation of ecotoxic effects was inconclusive. On the other hand, wetlands receiving 0.5 mg/L selenium from Kennecott Utah Copper near Salt Lake City some show no signs of toxicity (ept, 1997). Another study of an extensive wetland pond system in Benton Lake, Montana showed that inlet concentrations of 0.028 mg/L dissolved Se decreased to <0.00075 mg/L at the outlet, with no observable ecotoxic effects (Zhang and Moore, 1997). Based on this information, selenium at 0.010 mg/L is predicted to not exert effects in wetlands, indicating that constructing the South TMF Wetland to treat TMF outflow is acceptable.

The maximum flow discharging into the South TMF Wetland is predicted to be 0.44 m<sup>3</sup>/sec. This value will be used as the design flow rate for the South Treatment Wetland.

These metal loads to be removed are calculated by subtracting their concentrations in the TMF discharge by their Guideline values and multiplying this concentration by the design flow rate. The metal loads to be removed are presented in Table 17.

*Table 16. Metal loads to be removed from inflows to the South Wetland.*

Contaminant	Daily load to be removed
Cadmium	17.5 g/day
Copper	4.90 kg/day
Molybdenum	2.40 kg/day
Uranium	1.41 kg/day

Design removal rates were calculated above and are presented in Table 18 for each contaminant of concern. These daily metal loads are divided by these rates (Table 17) to determine the wetland surface area required for treatment.

*Table 17. Surface area required for the South Wetland.*

Contaminant	Areal Removal Rate	Surface area required
Cadmium	100 g/ha/day	0.18 hectare
Copper	1 kg/ha/day	4.9 hectare
Molybdenum	1.14 kg/ha/day	2.1 hectare
Uranium	0.23 kg/ha/day	6.1 hectare

Based on the above calculations, a 6.1 hectare wetland that receives drainage from the TMF will remove all its contaminants to CCME Guideline concentrations. The location of the South TMF Wetland is shown in Figure 1, above the TMF dam. This area is approximately 20 hectares, which is more than adequate to provide the 6.1 hectares required for treatment. Its discharge will flow directly to Casino Creek.

Note that all of the above removal rates were based on seasonal – including winter – treatment in cold climates. Thus, the South TMF Wetland should be able to continually treat discharge from the TMF pond during all seasons.

Although the South TMF Wetland is designed to remove metals from the TMF outflow, not selenium, it is possible to determine selenium removal rates and predict its concentration in the wetland discharge (i.e., concentrations entering Casino Creek).

Unfortunately, there are no treatment wetlands that provide useable information from which to derive removal rates and/or design criteria. Chevron's Richmond refinery treatment wetland generally decreases selenium concentrations to below detection limits, which means that it is loading-constrained. Selenium removal rates derived from their data will underestimate true removal rates.

Selenium fluxes across the water-sediment interface have been published for aquatic bodies in California (Oremland et. al., 1990), the Benton Lake wetland in Montana (Zhang and Moore, 1997), and the Goddard Marsh and Fording River Oxbow in BC (Martin et. al., 2008). The rates measured by Oremland and co-workers varied 0.006 to 22.1  $\mu\text{mol/L/hr}$  in sediments from different sites (Table 19). The authors calculated that these volumetric selenate reduction rates can be converted into areal rates of approximately 2.0-43  $\text{mg/m}^2/\text{day}$  (excluding the lowest rates). Rates measured in sediments from the Benton wetland were similar, though in the lower range. Selenium fluxes measured for the Goddard Marsh and Fording River Oxbow were reported to be 0.037 and 0.031  $\text{mg/m}^2/\text{year}$ , respectively. The latter rates appeared to be loading-constrained because of the low input selenium concentrations.

Taking into account the temperature differences between these various sites, the rate of selenium removal in the South Wetland is predicted to be 0.20  $\text{mg/m}^2/\text{day}$ . For an eight hectare wetland, this means that 16 g Se/day would be removed. At the design flows, the maximum daily selenium loading is predicted to be 209 g/day. Thus, only 8% of the selenium load would be removed by the South Wetland, decreasing its maximum concentration to 0.0051 mg/L in the wetland discharge. However, the data in Figure 13 indicate that such a concentration will still be protective in Casino Creek.

*Table 18. Rates of selenate reduction in sediments sampled from different sites.*

<b>Site</b>	<b>Selenate reduction (<math>\mu\text{mol/L/hr}</math>)</b>
Massie Slough	22.1/10.7
Big Soda Lake - littoral sediments	3.57
Lead Lake	3.01
Searsville Lake	1.91
Hunter Drain	0.74
June Lake	0.51
San Francisco Bay	0.41
San Joaquin Valley Pond	0.37
Big Soda Lake - pelagic sediments	0.21
San Francisco salina	0.12
Mono Lake - pelagic sediments	0.07
South Lead Lake	0.006
Roadside salina	<0.01

Taken from Oremland et. al., 1990.

Taken together, the bioreactor and the two treatment wetlands will remove every contaminant in drainage (except selenium) from the closed mine to CCME Guideline levels for protection of freshwater aquatic life. Moreover, these passive treatment systems will provide year-round protection, though the pit lake will only be discharged during the summer.

## ANTICIPATED TESTWORK AND PATH FORWARD

The above designs are predicated on a number of assumptions that must be confirmed during mine life. The contaminant loadings, especially from the HLF, pit walls, and TMF will need to be defined more precisely, as currently values are predicted based on modelled flows and chemistry. These predictions need to be confirmed by monitoring programs that are enacted during mining operations.

The bioreactor design will need to be validated through a series of trial runs of bench- and pilot-scale systems. First, a combination of laboratory and column studies will be used to establish fundamental metal removal rates at different retention times, determine which treatment processes play a role in metal removal and determine the optimum matrix mix for metal removal and maintenance of hydraulic performance. The two-stage treatment described earlier will be replicate in this column study. The composition of the matrix mix should only include materials that are available for full-scale construction. These materials should be tested for possible release of contaminants when the ambient redox potential becomes reducing. The benefits of feeding liquid organic carbon should also be evaluated and dosage should be optimized. Any requirement for pre- or post-treatment, such as TSS removal or aeration, should be determined during the column study. For this test work, partly detoxified process solution will be used to simulate draindown. If necessary, this water will be fortified with ammonium and/or nitrate at the concentrations predicted in draindown.

The above information will be used to determine a preferred design that will be demonstrated on a field-scale. This pilot trial should be run at approximately one-tenth of the design flows, i.e., 200-300 L/min and should use the optimal matrix mix and retention time. The trial should be run for approximately 6 months so as to provide data on seasonal performance. Inlet and outlet structures and the flow distribution network need to be designed so as to provide the information needed to engineer their full-scale counterparts. Similarly, construction methods that would be used for a full-scale facility (i.e., matrix mixing, placement of liner, etc) should be explicitly tested. This bioreactor should be instrumented to monitor internal processes, such as development of reducing conditions or heat loss. Tracer studies should be conducted to calculate the actual retention time and determine the discrepancy between predicted and actual. If liquid organic carbon is supplied, its consumption should be determined. Finally, a mass balance should be calculated after the small-scale bioreactor is deconstructed and sludge accumulation has been characterized. This will provide information on the longevity of the matrix mix and determine the fate and long-term stability of contaminants retained in the bioreactor. In addition, the fate of excessive BOD and/or hydrogen sulphide in the bioreactor discharge that is removed will be established and provide a basis for their management.

The longevity of the matrix mix will need to be assessed during the pilot trial, so that the full-scale bioreactor will be engineered with a specified design life. Typically, the design life for such a matrix will be 25-30 years, though this obviously depends on contaminant loading. The bioreactor matrix is expected to operate for approximately 5 years during HLF draindown, whereupon operation will be stopped and the matrix will be sealed in place. A



program to monitor residual accumulation in the matrix should be developed for full operation to confirm this presumed operating life.

The wetland design is based of empirically-derived metal removal rates measured at other mine sites. It will be necessary to obtain site-specific removal rates to size the North and South TMF Wetlands more accurately. Pilot-scale wetlands should be constructed early during mining operations and tested using mine water with chemistry close to that predicted at closure, such as the current drainage in Proctor Gulch.

The pilot wetlands should be sufficiently large (e.g., 1,000 m<sup>2</sup>) and mature (e.g., after vegetation is established, which takes 1-2 years) to provide information that can be used to design a full-scale system. The wetland cells will be vegetated with native plants (e.g., sedges such as *Carex aquatilis* or *rostrata*) obtained from a donor site or grown on site and planted as seedlings or rootstock. Plants will be selected according to their ability to grow in this region, overall robustness, tolerance of anticipated water depth and water fluctuations, and ease of transplantation on a large scale. Planting methods will be developed to account for the short growing season and large surface area that will need to be planted in a full-scale wetland.

The wetland will need to be operated for at least a year, receiving mine drainage with chemistry that bracket the potential range of metal concentrations predicted at closure. The pilot will be tested to determine treatment performance (i.e., how low can metal concentrations be decreased in the wetland), areal metal removal rates and to determine a design for either seasonal treatment (if possible) or year-round treatment. Different flow rates will be tested to determine the relationship between hydraulic retention time (HRT) and treatment performance. Actual HRT should be confirmed by conducting tracer studies.

Monitoring programs should be established to determine inflow and outflow water flows and chemistry, with an emphasis on metals and other contaminants of concern. Monitoring wells will also be installed inside the wetland to determine key operating parameters, such as water pH, oxidation-reduction potential and temperature. Changes in these parameters will be related to changes in operating conditions, such as flow rates, depth of water column or temperature. This information will be used to size and design the full-scale treatment wetland and determine optimal design parameters.

The wetland design will need to reflect that anticipated for the full-scale system. It is expected that shallow side slopes will be used and with fluctuating water levels. The pilot wetland should reflect this design and be similarly operated with fluctuating water flows and depths.

The fate of removed metals in this pilot wetland will be investigated to confirm that they are immobilized in wetland sediments and not taken up by wetland plants. As with the bioreactor trial, mass balances will be developed by sampling wetland sediments and determining the mass of metals retained in them. To that end, accurate and reliable flow meters will need to be operated at the wetland inlet and outlet.

Once the design criteria and wetland surface area/volume have been refined, the mine should confirm and refine the conceptual design presented above. Additionally, the full-scale design will need to account for other site-specific factors, such as winter operation, fluctuations of water depth, etc. In this regard, the lessons learned at Keno Hill, in the NWT, Nunavut and Western United States for comparable treatment systems will need to be applied in this design. A recent development in wetland design for cold climates is to evaluate wetland configurations that minimize heat loss from the system. This includes incorporating deep substrates, planting vegetation that forms dense clumps that form insulating layers by trapping snow and ice, etc. Although these systems operate with lower removal rates, their performance is often adequate because of low winter flow rates.

The conceptual design will be integrated in the design and evolution of the TMF and will need to consider long-term operation and maintenance issues for both facilities. Although wetland vegetation is not expected to require harvesting, there will be a need to maintain berms and flow control structures, repair damage caused by wildlife and restore hydraulic performance that may be altered by their activities (e.g., beaver dams). Access to the wetland and development of staging areas will be key elements of this.

Finally, the conceptual designs should identify design elements that promote their long-term integrity and ecosystem health, while maintaining their primary function as treatment wetlands.

The above tasks will help to refine the estimated capital and operating costs for the wetlands and may affect the amount of security required by regulators.



## CONCLUSIONS

This review and evaluation shows that a bioreactor and two treatment wetlands will remove the cadmium, copper, mercury, molybdenum, selenium, silver and uranium present in mine discharge and mitigate their potential impacts in Casino Creek. This conclusion is based on the predicted worst-case scenario concentrations for these metals in the heap draindown, pit lake discharge and TMF outflow for their respective maximum flows. Their effluents will meet CCME Guidelines for all these metals, though not for selenium discharged into Casino Creek. It is argued that CCME Guidelines for Protection of Freshwater Aquatic Life are inappropriate for this constituent and that selenium concentrations of  $<0.005$  mg/L in the treated TMF discharge are still protective.

The 57,000 m<sup>3</sup> bioreactor will be operated during draindown of the HLF, prior to draindown water being pumped to the open pit. It is planned to supply liquid organic carbon as the bioreactor is operated. Following draindown, the bioreactor will be shut down and the matrix will be permanently sealed in place.

A minimum seven hectare North TMF Wetland and six hectare South TMF Wetland will be constructed at the upstream and downstream ends of the TMF to remove metals from the pit lake and TMF discharges, respectively. Their surface area requirements were determined from empirically-derived metal removal rates for several natural and engineered wetlands reviewed in this assessment. Although some professional judgment is required in evaluating these different removal rates, they provided a reasonably consistent database from which to derive conservative metal removal rates that were applied to size these wetlands. Cadmium loadings were light in both wetlands and did not substantially affect the sizing requirement. Loadings of copper, molybdenum and uranium were more significant and presented roughly comparable surface area requirement for a treatment wetland. Note that the removal rates used in sizing these wetlands were conservative and based on the performance of wetlands operating in cold climates. As such, the North and South TMF Wetlands are sized conservatively and will have sufficient capacity to decrease inflow metal concentrations below CCME Guidelines under all circumstances expected at the site.

Although it is planned to operate the North TMF Wetland during the summer months, data from a number of natural wetlands shows that metal removal can also occur during winter months. This provides an assurance that waters downstream of the project area will be protected under all circumstances. Conversely, the South TMF Wetland is expected to operate year-round, as water upwells through the TMF. The design of this wetland will provide a means to capture this water throughout the year and remove its metal loads. The wetland configuration will permit operation during winter months, when it is covered with snow and ice.

While it is planned that the bioreactor will be supplied with liquid organic carbon during operation, both wetlands will be operated passively, without any operator intervention. However, all three treatment systems will be monitored during operation and have plans for their maintenance.

The approximate locations for the bioreactor and wetlands are shown in Figure 1. Their final dimensions, configurations and operating parameters will be determined through investigations that refine the anticipated flows and chemistry for the drainage requiring treatment, as well as testwork described above. This testwork includes bench-scale and field-scale trials that are operated for 1-2 years. These tests should be conducted during the operation of the mine or shortly after closure. These investigations will also include surveys to identify suitable construction material and vegetation for the wetlands. Methods to propagate and plant vegetation for the full-scale wetlands will also be determined.

The fact that natural wetlands remove the above contaminants shows that this is a naturally-occurring process that only needs to be reproduced in an engineered facility. Removal of these metals has been shown to occur through adsorption onto organic matter, conversion to non-mobile species and reaction with biogenic sulphide in the wetland sediments. These processes will be active in the North and South TMF Wetlands if properly designed, since organic matter will be produced abundantly and sulphate is present at adequate concentrations in the pit lake and TMF discharges.

These investigations also showed that metals retained in wetlands are not taken up in plant tissues, but “age” into gradually biologically unavailable forms in their sediments. This gives confidence that the wetlands will be able to function as a large, healthy ecosystem that supports mobile wildlife and resident populations, without impairment from metals accumulating in their sediments.

Selenium present in TMF drainage is predicted to exceed CCME Guidelines. Arguably, these guidelines are inappropriate because dissolved concentrations are poor predictors of ecotoxic effects for this metalloid. This fact is recognized by the US EPA as it is updating regulations for selenium. Data presented in the above discussion indicate that selenium does not accumulate in fish tissues at concentrations  $<0.018$  mg/L, implying that this is a safe selenium concentration for Casino Creek. Given that selenium concentrations in TMF drainage are predicted not to exceed 0.0050 mg/L, it is concluded that selenium will not exert ecotoxic effects in the creek and does not require treatment.

Aside from selenium, all the metals present at elevated concentrations in the TMF pond and pit lake drainage will be removed below CCME Guidelines. This will help ensure that the project does not cause any ecotoxic effects in the receiving environment.

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