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An integrated, risk-based approach for design of mine waste long-term disposal facilities

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Abstract

Base metal mines can produce large quantities of waste in the form of tailings and sludges which contain metals in various compound forms. Although, the waste may be neutralised before disposal, it can have high acid generating and metal leaching potential and therefore it is important to determine optimal treatment/mitigation/disposal methods and their associated risks in order to protect human health and the environment. A risk-based approach is proposed to determine the optimal disposal methodology for mine waste. The main steps include: hazard identification, characterisation, geochemical transport modelling, exposure effect modelling, risk estimation/characterisation and risk management. To demonstrate the applicability of this method, a case study illustrating four mine waste disposal options with three potential sources of Contaminants of Concern (COC) are considered. Using the selected COCs, the human health and ecological risk is evaluated against acceptance criteria for each design option. A Multi-Criteria Decision Making (MCDM) analysis framework is then used to optimise the waste disposal options based on criteria which includes risk, costs and environmental protection.

1 Introduction

Ecological or human health risk assessment is a common approach to derive environmental quality criteria or to serve as a basis for remediation decisions. However, a risk-based approach to waste management is not often employed at the design stage of a project. This work proposes a methodology for employing a risk-based approach to mine waste disposal management. This approach could also be applied to industrial waste or mining-related waste. The advantage of such an approach is the reduced long-term costs and liability of a project and the reduced environmental effects. Waste management involves balancing competing objectives of minimising risks and waste management costs within the constraints of the project. In general, the lower the risk level the higher the costs involved and vice versa. Asante-Duah (1993) describes an optimum combination of risk level and cost for a set level of acceptance. Other relevant work includes a risk assessment approach employed by Volosin et al. (1997) in the remediation of acid rock drainage and a risk-based assessment of soil and groundwater quality relative to different remediation strategies (Swartjes, 1999). Bonano et al. (2000) considered risk assessment in the decision analysis of environmental remediation alternatives; while database uncertainty was investigated by Nitzche et al. (2000) in reactive transport modelling through Monte Carlo simulations.

The approach in this work involves characterisation of mine waste, then employing this data in a contaminant fate and transport model to determine contaminant concentration levels, then exposures to receptors for selected COCs. A probabilistic approach is used to estimate the human health and ecological risk to receptors based on exposures due to different waste disposal options. Finally, a Multi-Criteria Decision Making (MCDM) methodology integrates risk with other disposal criteria to determine the most effective mine waste disposal systems through use of a case study (Figure 1).

2 Case study

This case study provides an example of a methodology to assess mine waste disposal methods at the design stage of a project. It does not represent a particular site location and results cannot be used to infer assessment of a specific location or waste but rather used as an application of the described methodology and all values used in reference to the mine waste and the site are for solely for illustration purposes. Results from any risk-based decision making process are site specific thus results will change with site location and waste characteristics.

2.1 COC identification

To identify the potential COCs the results of solid mine waste assay are compared with the Canadian Council of Ministers of the Environment (CCME) Soil Quality Guidelines (SQG) (CCME, 1999) and liquid mine waste assay compared with CCME Freshwater Aquatic Life (FAL) guidelines (CCME, 2003) as well as background and baseline data. For this mine waste, prior to treatment the liquid waste constituents that exceed one of the guidelines include: aluminium, nickel, copper, lead, selenium, and cadmium. Based on previous mine waste assays and our analysis the following metals or compounds exceed the CCME SQG: nickel, copper, cadmium, chromium and selenium. It was noted that copper, nickel and lead have the highest percent exceedance of the FAL guideline. The mine waste contains a high percentage of sulphur thus there is potential that the sulphur could oxidise and form acid rock drainage (ARD) causing leaching of metals from the waste or bedrock. Although the waste will be neutralised before it is sent for disposal, pH is considered a COC for the ecological risk assessment. From a comparison of the assay results on solid and liquid mine waste with guidelines, background and baseline concentrations the potential COCs selected were nickel, copper, lead and pH.

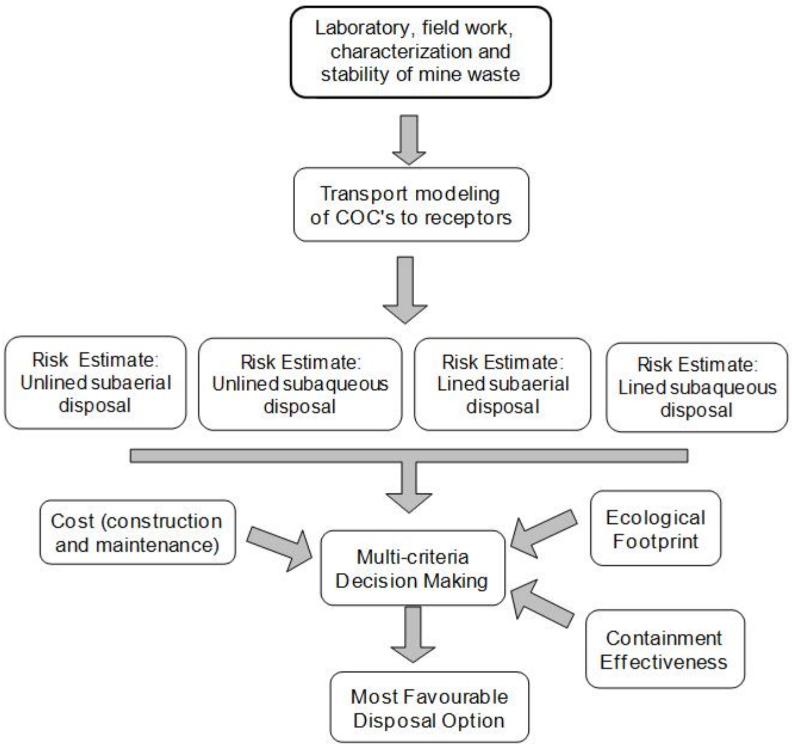

Figure 1 Schematic of study plan

2.2 Human health risk assessment

2.2.1 PCOC characterisation

The concentration of each COC is summarised in Table 1. Concentrations are provided for a representative mine waste impoundment decant water when it is neutralised and as a worst case scenario when it is not neutralised. The predicted COC concentrations in the groundwater at the base of the impoundment are provided for two main disposal options: 1) subaerial disposal 2) subaqueous disposal. All groundwater concentrations are based on values derived from reduced-scale field conditions and numerical modelling.

Table 1 Concentrations of COCs at source

COC	Decant Water – Neutralised (mg/L)	Decant Water – Not Neutralised (mg/L)	Groundwater[a] – Subaqueous (mg/L)	Groundwater[a] – Subaerial (mg/L)
	Subaqueous	**Subaqueous**	**Subaqueous**	**Subaerial**
Copper	0.01–0.14 (0.024)	0.47–1.59 (1.1)	0.01–0.03 (0.02)	0.02–1.09 (0.55)
Lead	0.002 (0.002)	0.002–0.037 (0.023)	0.002–0.003 (0.003)	0.002–0.016 (0.006)
Nickel	0.03–0.25 (0.11)	3.08–7.33 (5.2)	0.256–0.558 (0.4)	0.205–7.481 (3.5)
pH	7.1–9.7 (9.2)	2.8–6.4 (3.2)	9.2–9.8 (9.6)	3.1–4.2 (3.6)

Notes: a Groundwater concentrations taken at base of test disposal site. (…) average values.

Next the toxicity information for the identified COCs is determined for the various exposures. Of the COCs considered only lead is listed by the U.S. Environmental Protection Agency (U.S. EPA, 1991), as a probable human carcinogen (class B2). A chemical specific dose response relationship was used to characterise the health effects of lead. As a slope factor (SF) is not provided for lead by the U.S. EPA, a dose response relationship was derived based on results from a representative study on rats fed lead acetate or lead subacetate (U.S. EPA, 2006). The slope factor (SF) of $2x10^{-4}$ mg/kg/bw·d^{-1} was developed using the LMS model. The Reference Dose (RfD) for copper, lead and nickel were derived using data from US EPA Integrated Risk Information System (IRIS) (U.S. EPA, 1991), Agency for Toxic Substances and Disease Registry (ATSDR) (2004), U.S. EPA (2009) and Health Canada (2006).

2.2.2 Human health transport modelling and exposure modelling of PCOCs

To illustrate this methodology only receptors and exposure involving surface water and groundwater are considered; although exposure to COCs through dust or soil may be important for particular mine wastes and sites. At the Site the human receptors for fresh water could include fishers and swimmers. As a child is the most vulnerable receptor, for conservative analysis a child swimming in an offsite downgradient larger water body was selected as one receptor. A worker receptor at an industrial park, located downgradient from the disposal site, exposed to COCs through groundwater usage was selected as a second receptor. For this case study, two scenarios cause the transport of COCs from the mine waste impoundment to the receptor: dam overtopping and leachate migration. Receptor exposure is through 1) dermal absorption while swimming and 2) water ingestion during drinking and dermal absorption through showering using groundwater. For human health risk assessment lead and nickel are considered as COCs.

2.2.3 Human health risk scenario 1: exposure through dermal absorption during swimming

As a worst case scenario the concentration of metals in the stream during impoundment dam overtopping is equal to that in the decant water (Figure 2). A summary of the concentration of COCs in the larger water body at the site due to overtopping is provided in Table 2 along with water quality guideline and baseline concentration data. The COC concentration was determined for two sources; neutralised and not neutralised decant water (DW1 and DW2).

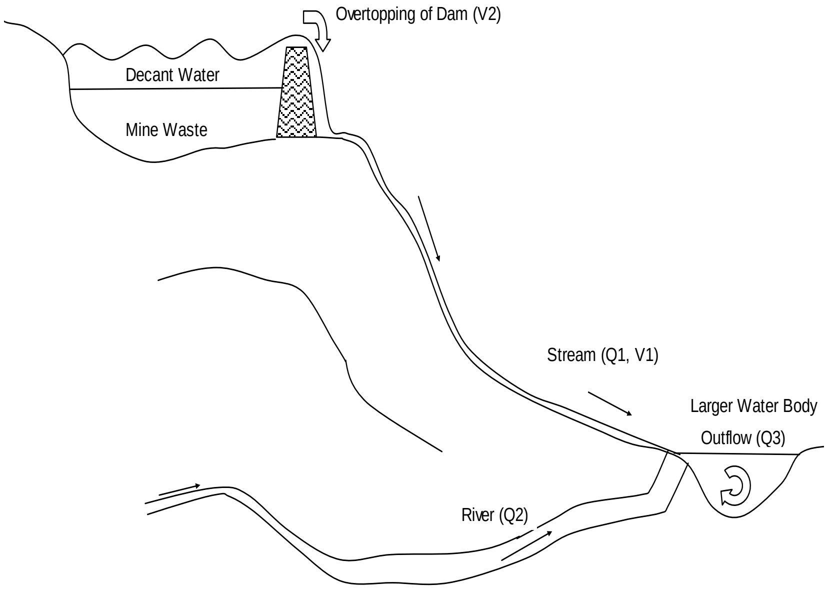

Figure 2 Schematic of dam overtopping and entering stream and larger water body

Table 2 Predicted metal concentrations in downgradient larger water body due to dam overtopping

COC	Water Quality Guideline[b] (µg/L)	Baseline Concentration[a] – Larger Water Body (µg/L)	Predicted COC Concentration in Larger Water Body (µg/L)	
			DW1[c]	DW2[d]
Copper	2	0.2–1.5	0.2–1.5	0.2–1.5
Lead	2	0.1–0.4	0.1–0.4	0.1–0.4
Nickel	8.3	<0.5	0.48–0.49	0.6–0.8
pH	NGA	NA	7.5–7.6	7.3–7.5

Notes: a. Ecological Risk Assessment for proposed development; b. Water Quality Guideline: B.C. MOE, 2006;
c. DW1: COC concentration at receptor with neutralised decant water;
d. DW2: COC concentration at receptor with not neutralised decant water; NGA: no guideline available NA: not available.

The larger body of water was modelled as both a continuous stirred tank reactor (CSTR) and a plug flow reactor. Although predicted concentrations are less than the water quality guideline; the risk to receptors is calculated to illustrate the methodology. As the calculated concentrations in the larger water body were similar for each source (DW1 and DW2) only one set of values were used as input for the exposure determinations.

2.2.4 Human health risk scenario 2: exposure through ingestion and dermal absorption of groundwater

Figure 3 illustrates the migration of COCs from the disposal site to the receptor at a proposed industrial facility through the groundwater. SESOIL (Yeh et al., 1987) combined with AT123D (Environmental Software Consultants (2006), is used to predict the migration of the leachate plume from the base of the disposal site to the receptor for both subaerial and subaqueous disposal cases. The advective-dispersive transport is described according to Robertson (1974).

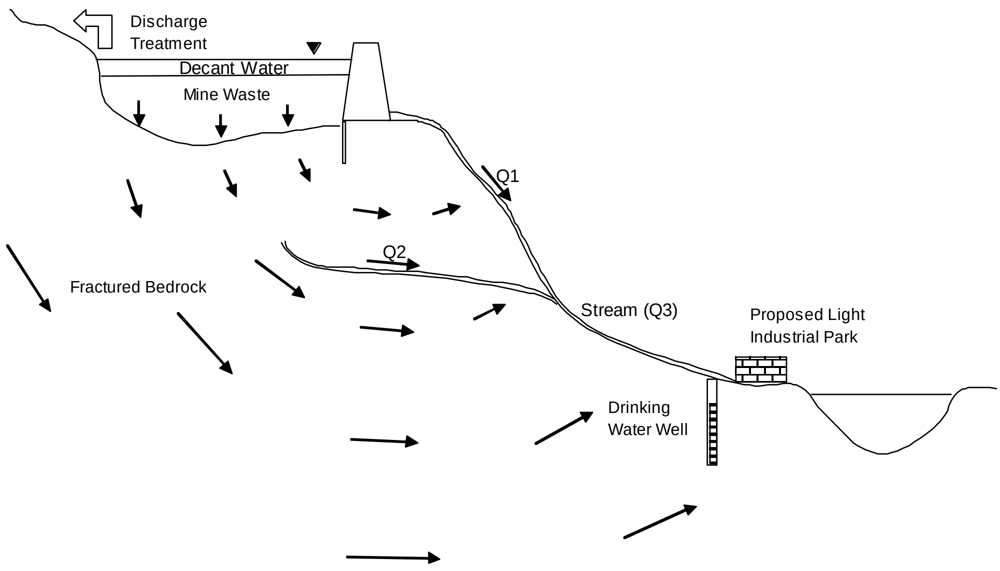

Figure 3 Schematic of leachate migration from disposal site into groundwater

Assuming a hydraulic conductivity of the bedrock of 5.0E–06 m/s, the maximum concentration of contaminant in the plume reached the receptor (1,200 m from the site) in between 145–151 years depending on the metal and the initial concentration. If the hydraulic conductivity was reduced to 1.0E–6 m/s the maximum concentration in the plume arrived at 600 m in 289 years and using 1.0E–05 m/s the peak concentration reached the receptor in 60 years. The arrival time is very sensitive to the hydraulic conductivity or extent of fractures in the bedrock which will in turn affect the concentration of contaminants at the receptors.

A summary of the concentration of metals in the groundwater due to leachate migration from the base of the disposal site for both disposal cases is provided in Table 3 along with the water quality guideline, background and baseline concentration data. The concentrations are derived for both the subaqueous and subaerial disposal cases. Although concentrations are below FAL guidelines receptor exposures are calculated to demonstrate study methodology.

Table 3 Predicted metal concentrations in groundwater due to leachate migration

COC	Water Quality Guideline (ug/L)	Background Concentration[a] (ug/L)	Baseline Concentration[a] (ug/L)	Predicted Groundwater Metal Concentration in Well (ug/L)	
				GW1-Subaerial	GW2-Subaqueous
Lead	1–7	<1	<0.5	1.3E–03 – 1.95E–03	1.3E–03 – 9.8E–03
Nickel	25	<1	<2–3.0	0.163–4.8	0.163–0.33

Notes: a. ERA for proposed development; GW1: COC concentration during subaerial disposal; GW2: COC concentration during subaqueous disposal.

Exposure Calculation: The exposures (chronic daily intake, CDI and lifetime average daily dosage, LADD) are calculated through equations (1), (2) and (3) (U.S. EPA 1988, 1989, 1992, 1997) for adsorption and ingestion using the values in Table 2 for skin absorption due to swimming and values in Table 3 for ingestion and skin adsorption due to showering. Where data are available exposure parameters are expressed as a probability density function (PDF); then Monte Carlo simulations are used through the code @RISK (Palisade Corp., 1991) to select parameter values and determine the cumulative density function (CDF) for the CDI and LADD for each COC. The exposure parameters are: concentration in water (CW), dermal absorption (DAevent), skin absorption rate (Kp), fraction absorbed (FA), surface area (SA), exposure frequency (EF), exposure duration (ED), event frequency (EV), body weight (BW), averaging time (AT), amount ingested (IR), bioavailability (ABSs) and fraction ingested (FI).

$$CDIorLADD_{absorption} = (DA_{event} \cdot SA \cdot EV \cdot ED \cdot EF) / (BW \cdot AT) \tag{1}$$

$$\text{DAevent} = \text{FA Kp CW} \tag{2}$$

$$CDIorLADD_{ingestion} = (CW \cdot IR \cdot FI \cdot ABs \cdot EF \cdot ED) / (BW \cdot AT) \tag{3}$$

2.2.5 Human health risk estimation

Using equations (4) and (5), CDI and LADD exposure functions and the COC RfD and SF values from toxicity data; values for hazard index (HI) and excess carcinogenic risk are derived for four mine waste disposal options: 1) subaerial and lined; 2) subaqueous lined; 3) subaerial and unlined and 4) subaqueous and unlined (Table 4 and 5). For this exercise a simplified assumption was made that the lined ponds are assumed to be leak proof. To account for variations in the dose response test results, RfD values for lead (0.0036 mg/kg bw/day) and nickel (0.02 mg/kg bw/day) and the SFlead (2.0E–04 mg/kg/ bw·day^{-1}) were described by a normal distribution. The range of values provided in Tables 4 and 5 are the 95% and 5% confidence limits of the CDF along with the 50% value of the CDF which are derived through Monte Carlo simulations with the code @RISK (Palisade Corporation, 1991). A sample plot of an HI CDF is provided in Figure 4.

$$\text{Total Hazard Index} \quad HI = \sum_{i}^{n} (CDI_i / RfD_i) \tag{4}$$

$$TotalCarcinogenicRisk = \sum_{i}^{n} (LADD_i \cdot SF_i) \tag{5}$$

Table 4 Hazard indices for COCs lead and nickel and select disposal methods

Disposal Method	Exposure Route	Lead[a]	HI Distribution[b]	Nickel[a]	HI Distribution[b]	Total HI
Subaerial – lined	Assuming no leakage and no decant water	-		-		0
Subaqueous – lined	Swimming	3.8E–09	1.8E–09–8.0E–09	5.1E–08	3.3E–08–7.5E–08	5.5E–08
Subaerial – unlined	Ingestion (groundwater)	3.8E–05	1.3E–05–9.5E–05	2.7E–03	0.3E–03–8.2E–03	2.7E–03
	Showering (groundwater)	1.9E–10	0.6E–10–4.8E–10	3.4E–07	0.46E–07–9.7E–07	
Subaqueous – unlined	Ingestion (groundwater)	1.6E–05	1.1E–05–2.0E–05	4.1E–04	2.7E–04–6.0E–04	4.3E–04
	Showering (groundwater)	8.0E–11	4.9E–11–10.3E–11	5.1E–08	3.0E–08–7.8E–08	
	Swimming	3.8E–09	1.8E–09–8.0E–09	5.1E–08	3.3E–08–7.5E–08	

Notes: a) 50 percentile value, b) range 5–95 percentile.

All of the HI values were well below the generally used reference value of 1.0 and the excess carcinogenic risk was significantly less than 1x10^{-6}. As the lined impoundments were assumed to be leakproof; leachate migration did not occur in this case resulting in no exposure to receptors via this route. The disposal options in order of lowest HI and carcinogenic risk values to highest were: lined subaerial, lined subaqueous, unlined subaqueous and unlined subaerial. The highest HI values were for nickel ingestion at HI equal to 2.7E–03 for subaerial and 4.1E–04 for subaqueous unlined cases.

The excess carcinogenic risk values were higher for subaerial unlined disposal than subaqueous unlined. The highest carcinogenic risk value was that for ingestion of groundwater at 1.1E–11 in the subaerial unlined case.

Table 5 Carcinogenic risk for COCs lead and nickel and select disposal methods

Disposal Method	Exposure Route	Carcinogenic Risk (Lead)[a]	Distribution[b]	Total Carcinogenic Risk (Lead)
Subaerial – lined	Assuming no leakage and no decant water	0.0E+00	–	0.0E+00
Subaqueous – lined	Swimming (marine)	1.9E–16	0.92E–16 – 3.9E–16	1.9E–16
Subaerial – unlined	Ingestion (groundwater)	1.1E–11	0.35E–11 – 2.8E–11	1.1E–11
	Showering (groundwater)	5.5E–17	1.7E–17 – 13.0E–17	
Subaqueous – unlined	Ingestion (groundwater)	4.7E–12	3.1E–12 – 5.6E–12	4.7E–12
	Showering (groundwater)	2.4E–17	1.6E–17 – 3.4E–17	
	Swimming (marine)	1.9E–16	0.92E–16 – 3.9E–16	

Notes: a) 50 percentile, b) range 5–95 percentile.

2.2.6 Uncertainty in human health risk assessment

For human health risk assessment, the contribution to the uncertainty was evaluated for select parameters for each pathway using the spearman rank. From these results the COC concentration (CW) is the most dominant factor for most of the lead and nickel exposures for HI and carcinogenic risk for all three pathways. There has not been an attempt made to address uncertainty in all the exposure parameters. Examples of a few other influences on results include: site location; waste type; bedrock type, permeability and fracturing; subsurface and surface water chemical reactions; and liner permeability.

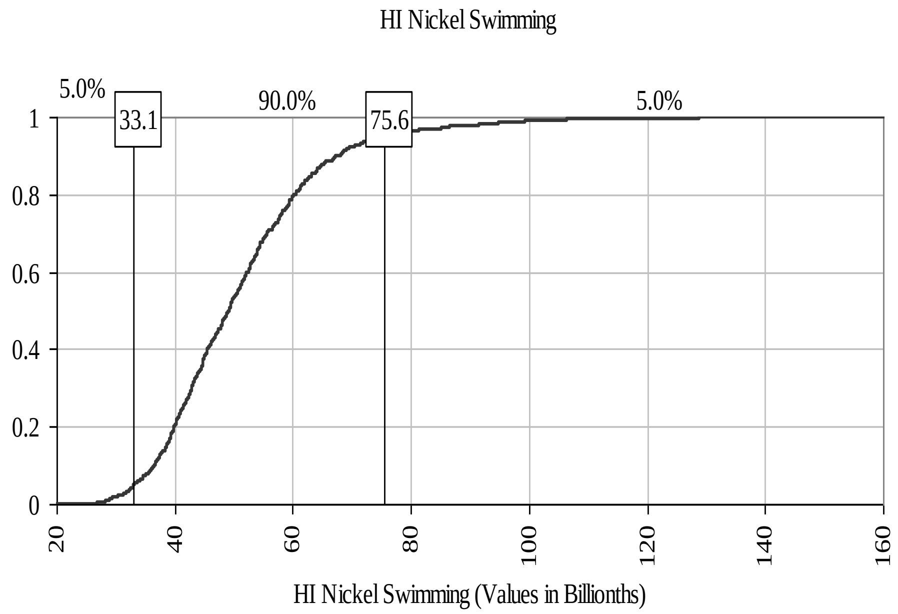

Figure 4 CDF of HI for nickel with swimming pathway

2.3 Ecological risk assessment

2.3.1 COC identification and characterisation

For this study the effect of COC concentration on rainbow and brook trout in the site stream and in an off-site larger body of water was assessed. The two sources of COCs were mine waste impoundment decant water through dam overtopping and mine waste leachate through groundwater migration. Rainbow trout and brook trout were selected as the valued ecosystem components (VECs) for the freshwater environment. No observed effects concentration (NOEC) data from the U.S. EPA ECOTOX database (U.S. EPA, 2006) for the aquatic species and metal speciation of interest was selected and plotted as a PDF. From the CDF of the NOEC data the 5 percentile exceedance value is derived for the species and metal and compared with CCME guidelines. The estimated threshold reference value (TRV) was selected from the lower of that determined from NOEC values and the CCME FAL guideline.

2.3.2 Ecological transport modelling of COCs

2.3.2.1 Ecological risk scenario 1 a) and b): exposure in on-site stream from dam overtopping and leachate migration

Figure 2 illustrates the overtopping of the impoundment dam by decant water and the stream immediately downgradient of the impoundment. For the case of dam overtopping, the COC concentration is equal to that of the impoundment decant. For the case of leachate migration, the leachate migrates through the mine waste and into the groundwater in the bedrock then the regional groundwater flow system disperses and transports the leachate in the direction of groundwater flow (Figure 3). For the site stream, groundwater is a major contributor to its discharge. As leachate from the base of the impoundment enters the groundwater it contributes to the base-flow of the stream. The COC concentration in the stream due to leachate in the groundwater is determined based on the contribution of baseflow to the overall stream discharge. A summary of the concentration of COCs in the stream due to dam overtopping and leachate migration is provided in Table 6 along with the water quality guideline and groundwater baseline concentration data. Predicted average copper, nickel and lead concentrations in the stream due to dam overtopping (with decant water neutralised and not neutralised) and leachate migration (for the subaerial and subaqueous case) exceed the CCME water quality guidelines (CCME, 2003). The pH measurement for the COC sources of: DW2 (not neutralised) and GW^{-1} (subaerial) had pH ranges outside that of CCME guideline for freshwater aquatic life (CCME, 2003).

Table 6 Predicted metal concentrations in stream due to dam overtopping and leachate migration

COC	Water Quality Guideline (µg/L)	Background Concentration[a] (µg/L)	Baseline Concentration[a] (µg/L)	Predicted Stream Metal Concentration (µg/L)			
				Dam Overtopping		Leachate Migration	
				DW1	DW2	GW1	GW2
Copper	2	<1–2 (1.1)	<1–14 (2.1)	7.6–110 (59)	360–920 (640)	6.8–370 (190)	3.4–19 (11)
Lead	1	<1	<1–10 (1.75)	1.5–1.7 (1.6)	1.5–21 (11)	0.74–6.0 (3.4)	0.74–1.74 (1.2)
Nickel	25	<1	<1–3 (1.2)	23–190 (110)	2,300–4,200 (3,200)	69–2,500 (1,300)	8.6–190 (99)
pH	6.5–9	??	5.74	6.8–8.7	3.5–4.7	4.9–5.2	6.9–7.1

Notes: a. ERA for facility; DW1 source is neutralised decant water; DW2 source is not neutralised decant water; GW1: subaerial disposal values; GW2: subaqueous values; (...): 50 percentile value.

2.3.2.2 Ecological risk scenario 2: exposure in larger water body from dam overtopping

The concentration of metals in the larger water body was derived previously (Table 3). As is evident from Table 3 the predicted concentrations for copper, lead and nickel are below B.C. marine water quality guidelines (BC MOE, 2006) and the concentrations are very close to baseline concentrations in the larger body of water therefore these specific COCs, pathway and receptor are not considered further in this ecological risk assessment.

2.3.3 Risk estimation and uncertainty: ecological

To consider the uncertainty associated with the predicted stream COC concentrations each COC concentration range in Table 6 is described using the 95 % and 5 % confidence limits of the CDF lognormal distribution. The TRV values are described using a normal distribution. Using equations (6) and (7), values for ER and total ER are derived for the four disposal options: subaerial and subaqueous lined and unlined disposal options (Table 7). A sample plot of an ER CDF is provided in Figure 5. The 50 percentile ERs for rainbow trout and COCs copper, nickel and lead are derived using these distributions (Table 7) for each of the four disposal methods as shown in Table 7. Due to their ER values (greater than 1.0) copper, nickel, lead and pH could be brought forward for further assessment for freshwater aquatic life. In general, the disposal options in order of lowest to highest ER values for the COCs and pathways selected are: lined subaerial, unlined subaerial, lined subaqueous and unlined subaqueous. This ranking is based by the assumption that the lined impoundment does not leak. Except for the lined subaerial case, the ER values were greater than 1.0 for each disposal method, COC and both exposure routes. The highest ER values are for the COCs copper and nickel, the pathway dam overtopping with non-neutralised decant water. In addition, the scenarios of dam overtopping with non-neutralised decant water and leachate migration with subaerial disposal, are expected to have negative effects on aquatic freshwater life in the site stream due to low pH.

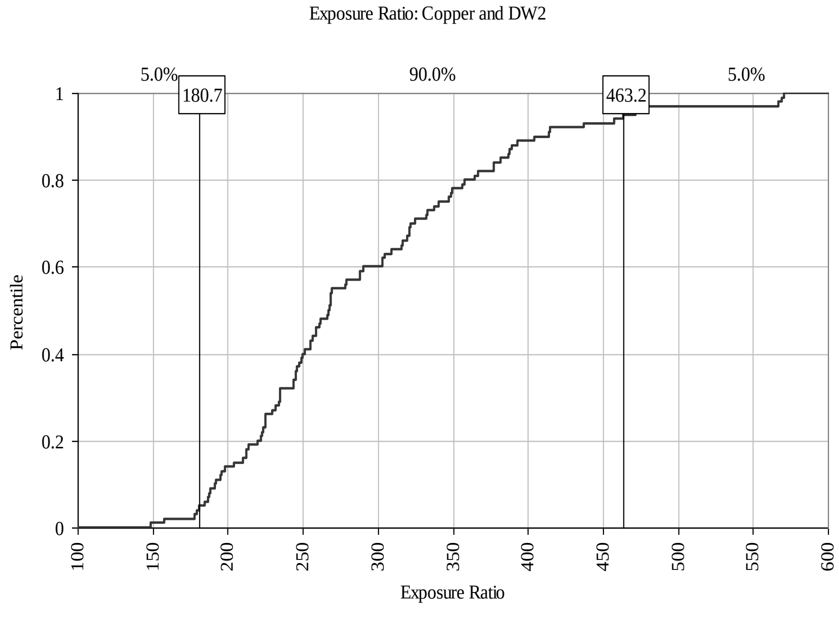

Figure 5 CDF of exposure ratio for copper in stream due to dam overtopping

$$\text{Exposure ratio (ER)} = \frac{\text{estimated exposure concentration (EEC)}}{\text{TRV}} \quad (6)$$

$$ER = \sum (EEC/TRV) \quad (7)$$

Table 7 Exposure ratios for COEPC's and rainbow trout on nearby stream

Disposal Method	Exposure Route	ER (50 Percentile Value)			ER	Total ER
		Copper	Nickel	Lead	pH	
Subaerial – unlined	Groundwater – GW1	50	55	2.3	>1.0	107
	Dam overtopping – DW1	18	6.5	1.6	<1.0	
Subaqueous – unlined	Dam overtopping – DW2	265	315	6.75	>1.0	37; 598
	Groundwater – GW2	4.4	5.3	1.2	<1.0	
Subaerial – lined	No leakage and no decant water	0	0	0	<1.0	0
Subaqueous – lined	Dam overtopping – DW1	18	6.5	1.6	<1.0	26; 587
	Dam overtopping – DW2	265	315	6.75	>1.0	

Note: Two values for ER are for calculations with either neutralised or not neutralised decant water.

3 Multi-criteria risk-based decision making

The decision hierarchy for this case study was provided in Figure 1 and includes the criteria of ecological and human health risk, cost (construction and maintenance), ecological footprint and containment effectiveness. Using the Analytical Hierarchy Process (AHP) methodology for optimisation seven alternative matrices, two pair-wise matrices, one goal matrix and a synthesis matrix were developed. The values (1–9) and reciprocals used to compare the alternatives are based on the authors judgment and used mainly for illustration purposes. From the results of the synthesis matrix (Table 8) which includes all decision criteria, the order of preference for disposal methods from highest to lowest is; lined subaqueous, lined subaerial, unlined subaqueous, and unlined subaerial.

Table 8 Synthesis matrix for optimal mine waste disposal method

Disposal Options	Human Health Risk		Cost		Ecological Risk	Containment Effectiveness	Ecological Footprint	Overall Priority
	0.347		0.098		0.225	0.173	0.156	
	Carcinogenic Risk	Non-carcinogenic Risk	Construction Cost	Maintenance Cost				
	0.667	0.333	0.167	0.833				
Unlined Subaerial	0.088	0.088	0.532	0.063	0.055	0.066	0.062	0.078
Lined Subaerial	0.213	0.213	0.099	0.25	0.166	0.404	0.205	0.235
Unlined Subaqueous	0.153	0.153	0.297	0.143	0.201	0.68	0.139	0.148
Lined Subaqueous	0.546	0.546	0.071	0.545	0.578	0.462	0594	0.538

4 Conclusions

A risk-based approach to decision making was used to assess disposal options for a typical mine waste. Mine waste characterisation data and contaminant fate and transport modelling predicted exposure to potential receptors. A probabilistic approach was then employed to estimate the risk to the receptors and its uncertainty based on different mine waste disposal options. Finally, multi-criteria risk-based decision making, which integrates risk assessment with other disposal criteria, was used to determine the optimal disposal option.

Three different disposal priority rankings were obtained for the mine waste depending on whether the ranking was based solely on human health or ecological risk or determined using a multi-criteria decision making process. The ecological risk had a different disposal ranking than the human health risk; due to the inclusion of risk to the VECs in the site stream and the dominant effect of the non-neutralised decant water. According to the predicted ER values for the site stream, the VECs would be affected both COC metal concentration and pH via leachate migration and dam overtopping. For this case study, in order to protect the stream it would be important to eliminate leachate migration from the mine waste impoundment. If subaerial disposal is an option, there is no risk of dam overtopping thus one less factor contributing to the total risk. Although not considered, it is anticipated that any subaerial disposal site will require a cover to protect the local environment from air transport of waste particulate. For this case study, leachate migration was not predicted to cause a significant risk for users of a downgradient well thus also not for exposures to an off-site larger water body. The actual ranking of disposal options is site and waste specific and can incorporate additional factors and decision criteria.

As indicated previously there are many other parameters that deserve more detailed consideration or preliminary consideration when evaluating disposal methods. They could include but are not limited to: bedrock type, permeability and fracturing; site location; waste characteristics; subsurface and surface water chemical reactions; leakage and degradation rate and type of liner system; and modelling of COC transport. For this case study, human health risk and ecological risk had the highest score of the five decision criteria in the MCDM analysis. This analysis helped to demonstrate the significance of these risks and the corresponding importance of the long-term integrity of the disposal site on the selection of an optimal waste disposal method.

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