
Elk Valley Water Quality Plan

Annex D.4

Geochemical Source Term Inputs and
Methods for the Elk Valley Water Quality
Planning Model

Teck

Geochemical Source Term Inputs and Methods for the Elk Valley Water Quality Planning Model

Prepared for

Teck Coal Limited

Prepared by



SRK Consulting (Canada) Inc.
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Executive Summary

Geochemical source terms were developed as inputs into the water quality planning model to support the Elk Valley Water Quality Plan. This report summarizes the method used to develop the geochemical source terms.

All source terms were derived starting from conceptual geochemistry based on current understanding of the geochemical characteristics of the mines waste and the performance of the disposal facilities.

For waste rock spoils, pitwalls, and co-disposed waste rock and coal rejects, releases of selenium, sulphate and nitrate were calculated based on the volume of material in each catchment, the release rate of each parameter derived from monitoring data and the predicted infiltration rate. Selenium and sulphate originate from oxidation of pyrite whereas nitrate is from leaching of explosives residuals. The method assumed that a significant proportion of spoils is active in generating soluble load. The resulting concentrations of selenium and sulphate derived by the method were constrained to not exceed maximum possible concentrations indicated by the solubility of gypsum, a secondary mineral expected to form in the spoils.

For nitrate, leaching rate decreases as a function of increasing age of spoil. Therefore, the source term was based on the average of the spoil. The resulting concentrations were not constrained due to the high solubility of nitrate.

The source terms for other elements (for example, cadmium) leaching from waste rock spoils were based on fixed concentrations indicated by monitoring data.

The source terms for spoils account for observed seasonal variation in loads released (greatest loads during highest flows generated by snowmelt) and response to differences in annual flows.

For coal reject disposal facilities, weathering and leaching occurs by similar processes as waste rock but it was observed that oxygen penetration is limited to a surface zone and that the cores of coal reject facilities do not oxidize. Therefore, the source term was based on fixed concentrations applied to all infiltrating water into the facilities rather than calculated from the entire volume of coal reject.

1 Introduction

1.1 Background

Teck Coal Limited (Teck), as part of its overall water quality management program, has the need to estimate current and future concentrations of selenium, sulphate, nitrate and cadmium in the Elk River resulting from operation of its five coal mines. Essential inputs into such estimation work are geochemical source terms for the chemical loadings into the water resources from mine facilities and workings.

SRK was originally requested to develop the geochemical source terms as part of water quality predictions for permit amendment activities related to expansion of mining at Elkview Operations (EVO) into Baldy Ridge. The method of estimating was subsequently updated and used to support environmental assessments for the Line Creek Operations (LCO) Phase II and Fording River Operations (FRO) Swift projects, and to support permit amendment applications the Greenhills Operations (GHO) West Spoil project. The source terms were also used in the water quality model that supported the valley-wide selenium management plan.

The same geochemical source terms are now being applied to the Elk Valley water quality model, which supports the development of the Elk Valley Water Quality Plan (the Plan). This report includes all input assumptions and relationships used in the model, and provides explanations of source term development and application.

1.2 Data Sources

Input data used to support source-term calculations were largely provided by Teck and are listed in Table 1.

Table 1. Information Used to Develop Inputs into the Source-term method

Information	Teck Coal Operation	Time Period Covered
Spoil volumes by operation	All	To 2011, depending on operation
Near-source surface water chemistry	All	To 2011, depending on operation
Near-source flows	All	To 2011, depending on operation
CCR humidity cell data	LCO, GHO	N/A
Waste solids chemistry	EVO, LCO, FRO, GHO	N/A
In-situ gas monitoring	GHO, LCO	2009 to 2012

Note: CCR = coarse coal reject

1.3 Structure of this Document

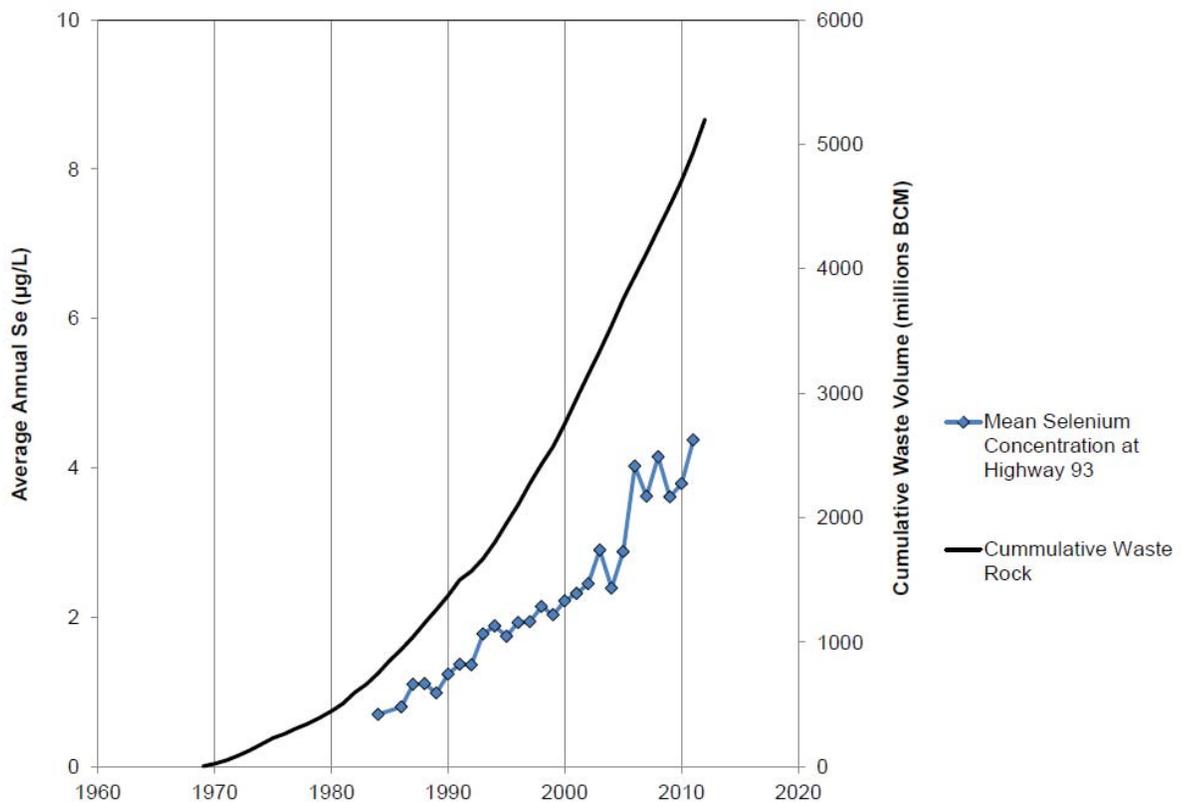
This report is structured as follows:

- Section 2 describes the conceptual geochemical model (CGM) for each facility requiring a source term and the mathematical implementation of the CGM.
- Section 3 describes data sources and analysis undertaken to develop the inputs into the geochemical source-term method.
- Section 4 provides discussion of various aspects of the geochemical source term calculations.

2 Geochemical Source Term Calculation Method

2.1 Overview of Methodology

The geochemical source-term method was selected to be consistent with conceptual models formulated in the late 2000s describing leaching of selenium and other constituents from mine wastes in the Elk Valley. The primary observation justifying the conceptual models was that selenium concentrations in the Elk River were increasing with waste rock accumulation (Figure 1), which implied that selenium release could be predicted from the waste volumes generated by future mining operations.



P:\02_MULTI_SITES\Elk_Valley_Coal_Corp\1CT017.054_Geochem Documentation for EV\WQP100_Report\Figures\Elk River Concentration

Figure 1. Mean Annual Selenium Concentrations (in µg/L) in the Elk River Compared to Spoil Volume in Millions of Bank Cubic Metres (BCM)

(Source: Teck Coal)

The geochemical source-term method for describing leaching of selenium and other constituents from mine wastes in the Elk Valley was developed to achieve two main objectives:

- To be generally consistent with predictive water chemistry methods used elsewhere in BC, so that the method used for the Elk Valley would be recognizable and generally accepted by other industry and regulatory practitioners in this field.

- To use data primarily from geologically analogous sites in the Elk Valley, with the intent of eliminating inherent limitations and conservatism resulting from determining field-scale release rates based upon laboratory test results.

The first objective was achieved by using a conventional methodology that links weathering rates of primary minerals to generation of soluble weathering products, and the finite solubility of the weathering products. This approach has been applied and accepted for numerous projects in BC (for example, Western Canadian Coal 2005; Western Coal 2010; SRK 2012).

The second objective was achieved by demonstrating the geological and geochemical consistency of the Mist Mountain Formation throughout the Elk Valley. Studies of selenium distribution by Lussier (2001), Ryan and Dittrick (2001) and Ryan et al. (2002), and studies subsequently performed to prepare for regulatory submissions by Teck have shown that selenium and sulphur concentrations range over similar values throughout the Elk Valley (Figure 2). Sulphide-sulphur concentrations are typically less than 0.2% and selenium concentrations vary from less than 1 mg/kg to typically about 5 mg/kg (with some exceptions). Selenium concentrations tend to be lowest in sandstones and highest in mudstones. These observations support the application of information obtained on geochemical characteristics at four operations (EVO, LCO, GHO, FRO) to expansion of the operations, or watersheds with less data, provided they are in the same geological setting. Coal Mountain Operations (CMO) has some structural and stratigraphic differences and, therefore, is considered separately.

To summarize the geochemical source-term method, the amount of waste generated per year is multiplied by the amount of soluble rock components generated per m³ (cubic metre) of waste to give the amount of soluble rock components generated per year. That quantity is then divided by the volume of water estimated to infiltrate the rock during the year, giving the concentration of soluble components. The resulting concentration is then compared to the solubility limit to determine if the calculated concentration needed to be constrained to the limit. For less soluble components for which experience suggests that release by weathering far exceeds solubility limits, the source term concentration is set to the solubility limit. The approach was modified slightly for nitrate released from explosives residue.

2.2 Scenarios

To provide an evaluation of prediction uncertainty and to align the geochemical source-term method with similar methods used elsewhere, inputs to the calculations were provided for two cases: the best-estimate case based on mean or median statistics of data distributions for rates and concentrations, and the reasonable worst case, based on 95% upper confidence limit on the mean, 95th percentile or maximum of a data distribution. The mean and confidence limits on the mean were used to define the range of release rates because the mean of monitoring data is expected to define a typical value for the Elk Valley. Percentile statistics (median, 95th, maximum) for monitoring data were used to constrain solubility limits, because solubility is most likely constrained by high values representing minimal dilution of source waters. The choice of 95th percentile or maximum depended on the size of the dataset. For small datasets, the maximum value was used.

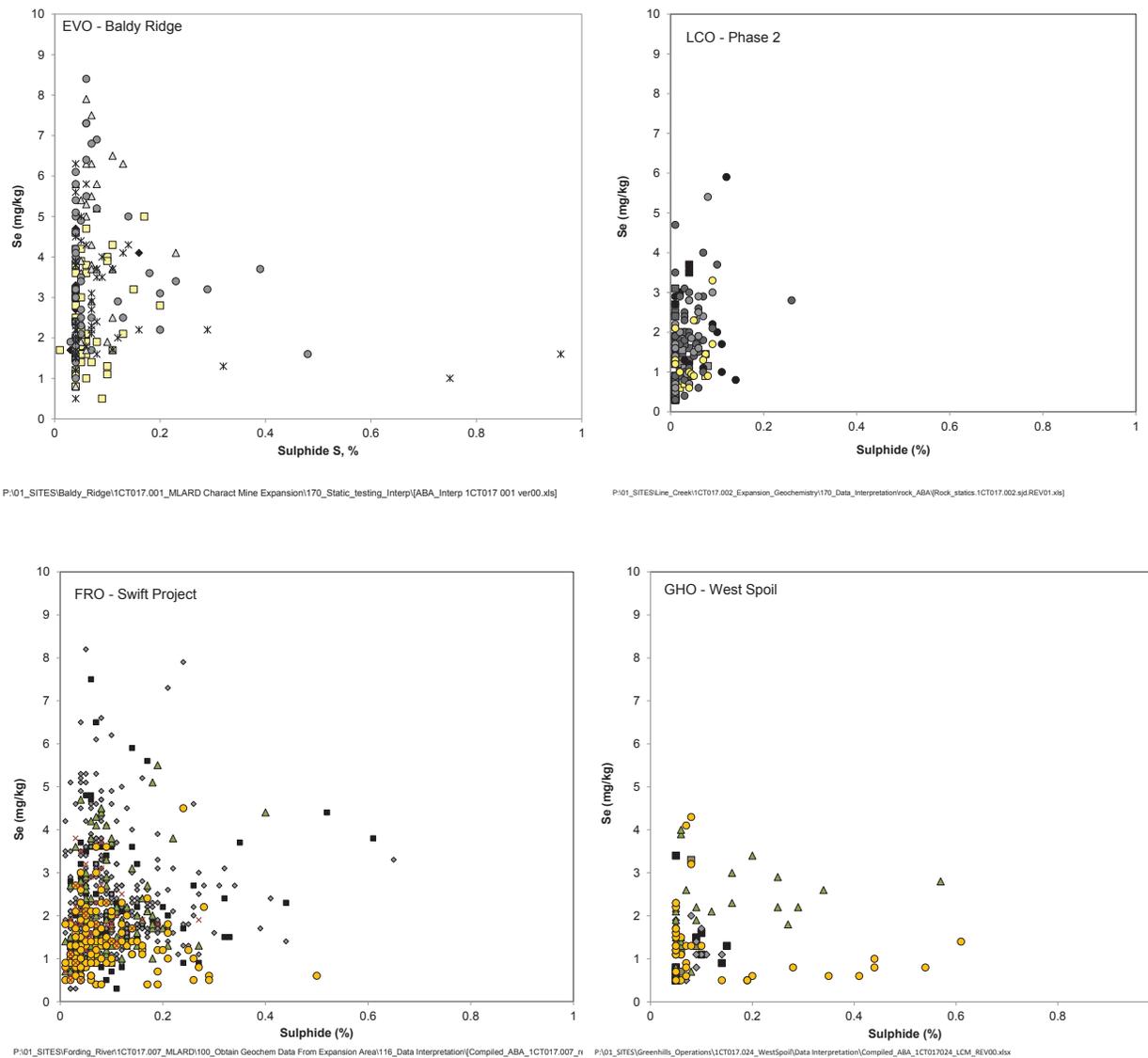


Figure 2. Selenium and Sulphur Concentrations in Rock Samples from the Mist Mountain Formation of Elk Valley.

Colours of symbols are rock types (yellows – sandstone, greys – siltstones and mudstones, black – coal)

2.3 Strengths and Limitations

The following are strengths of the method:

- The conceptual model used to develop the geochemical source-term method is based on a good understanding of the underlying weathering and leaching mechanisms.
- Coal is mined almost exclusively from a single geological formation, with what appears to be relatively uniform regional characteristics which allows the method to be applied throughout the Elk Valley.
- The conceptual model described in the following sections and used to develop the geochemical source-term method is primarily empirical rather than theoretical. An empirical method is considered better than a construct based on theory because it uses actual observations.
- Inputs to the method are based on a very large unique regional flow and water chemistry monitoring database accumulated by Teck over several years and more than a decade in some cases.
- Application of the method to valley-wide predictions of selenium, sulphate and nitrate has shown that predicted water chemistry aligns with observations.
- The geochemical source-term method developed from the conceptual model can be applied to historical wastes for which monitoring data are not available. However, if the method is applied retrospectively to sites for which site-specific monitoring data are available, factors must be applied to the predictions to calibrate the monitoring data to reflect local conditions. The benefit of using calibration factors is that they allow deviations from average valley-wide rates to be compared across sites, thereby facilitating evaluation of explanations for differences. As a potential future improvement to the method, monitoring data may be used to develop specific loading terms for the monitoring locations. Calculation of site-specific rates ensures that predictions will match observations, and may result in higher certainty for future predictions.

The following limitations of the geochemical source-term method should be considered:

- The method does not attempt to mathematically define geochemical processes but rather is an empirical tool to support simulation of downstream water quality concentrations. While some effort is made to achieve ion-balanced water chemistry, the method is largely focused on providing source terms for trace elements (e.g. selenium) rather than major ions (e.g. sulphate). The result of this limitation is that synergistic effects among ions may not be reflected. For example, elevated barium concentrations could be predicted with elevated sulphate concentrations, but, in reality, barium concentrations are suppressed by the low solubility of barite (barium sulphate).
- The geochemical source-term method was developed for application to future wastes for which drainage monitoring data are not yet available; but these wastes will mainly come from the Mist Mountain Formation, for which waste placement approaches are similar to elsewhere

in the valley. The method is not yet suitable for application to wastes from the Morrissey and Fernie Formations beyond where they occur at CMO; however, these have much smaller volumes than the Mist Mountain Formation.

- The method does not consider year-on-year lag effects. It is likely that there will be a lag in placement of wastes and contribution of the waste to chemical loadings. Consequently, the method may over-predict chemical loadings for new waste sources.
- The method is steady state for rock weathering components (selenium and sulphate), and does not consider factors that could cause long-term changes in chemical loadings, such as depletion of mineral sources.
- Since the method is empirical rather than mechanistic and based on existing monitoring data, the effect of source control measures, such as infiltration reduction by covers, cannot be readily assessed, and/or must use conservative assumptions to address uncertainty. As Teck develops information about the influence of covers on geochemical processes, the approach to estimating the influence of source control measures will be refined.

The method was developed before some of the supporting research related to the Plan was started; it therefore only partially incorporates ongoing findings of that research. Periodic reviews of the approach are planned to consider and incorporate findings of the program, particularly as they pertain to weathering, infiltration, leaching, and the performance of control measures. Moreover, monitoring over time will be required to confirm the implied linkage of tonnage to loading as mining methods and dump placement methodologies are refined in the future.

2.4 Implementation of the Source-Term method

The following sections describe the method used to calculate source terms for open pits and waste management facilities (i.e., waste rock dumps). Each section provides the conceptual model for the indicated source term as developed in 2010, which was then implemented in the calculation.

Table 2 provides a list of symbols used for the equations in the subsequent sections.

Table 2. List of Symbols

Symbol	Units	Definition	Assigned Values
J	None	Parameter (e.g. sulphate).	-
R_j	mg/m ³ /year	Rate of generation of parameter j from a bank cubic metre (Bank m ³) of waste rock	Table 3 and Table 4
$V_{rock}(t)$	bank m ³ (BCM)	Volume of rock placed at disposal location in year t	Mine plan
V_{rock}	bank m ³	Cumulative volume of rock at a disposal location	Calculated
\bar{t}	years	Average age of rock at disposal location	Calculated
$L_{i,j}$	mg/year	Initial calculation of load leached of parameter j from waste rock	Calculated
\bar{Q}	L/year	Average volume of water infiltrating the waste rock facility in a year	Infiltration estimate
Q	L/year	Volume of water infiltrating the waste rock facility in a year	Water balance
$C_{i,j}$	mg/L	Initial pore water concentration of parameter j	Calculated
C_{s,SO_4}	mg/L	Solubility limit for sulphate as defined by gypsum solubility	Section 3.1.4
$C_{s,Se}$	mg/L	Solubility limit for selenium assuming co-precipitation with gypsum	Section 3.1.4
$M_{Mg/Ca}$	mol/mol	Average molar ratio of magnesium to calcium	Section 3.1.4
$C_{s,j}$	mg/L	Constraining concentrations for parameter j	Section 3.1.4
$C_{c,j}$	mg/L	Calculated annual pore water concentrations	Calculated
$L_{a,j}$	mg/year	Annual load of parameter j calculated from $C_{i,j}$	Calculated
$p_{m,j}$	Unitless	Fraction of load of parameter j released in month m	Table 5
$L_{m,j}$	mg/month	Monthly load of parameter j calculated from $L_{a,j}$	Calculated
Q_m	L/month	Flow in month m	Water balance
$C_{m,j}$	mg/L	Monthly seepage concentrations	Calculated
$C'_{m,j}$	mg/L	Adjusted monthly seepage concentrations from Step 8 (See Section 2.5.2)	Calculated
$L_{m,j}$	mg/month	Monthly loads from Step 8 (See Section 2.5.2).	Calculated
C_{TDS}	mg/L	Calculated total dissolved solids concentrations	Calculated
$p_{uncontacted}$	unitless	Fraction of rock not contacted by meteoric water	0.5
$V_{flooded}$	bank m ³	Volume of rock inundated by water	Mine plan and water balance
t_{flood}	year	Time when rock is flooded	Mine plan and water balance
$T_{placement}$	year	Time when rock is inundated	Mine plan
$L_{uncontacted,j}$	mg	Load leached by flooding of rock volume not contacted by meteoric water	Calculated
$L_{contacted,j}$	mg	Load leached by flooding of rock volume contacted by meteoric water	Calculated
$L_{flush,j}$	mg	Total load leached by flooding of rock	Calculated
A_{wall}	m ²	Area of pitwall	Mine plan
D	M	Reactive thickness of pitwall	2
V_{wall}	m ³	Reactive volume of pitwall	Calculated

2.5 Permanently Exposed Waste Rock Source Term

2.5.1 Conceptual Model

The conceptual model for weathering of waste rock is well-established in the geochemical literature, though release of selenium may be affected by its specific chemical properties. Specifics of the model for coal wastes and leaching of selenium have been defined previously (e.g., SRK 2008) and refined through ongoing efforts (e.g., SRK 2013b).

Weathering and leaching of waste rock occurs predominantly through several groups of processes:

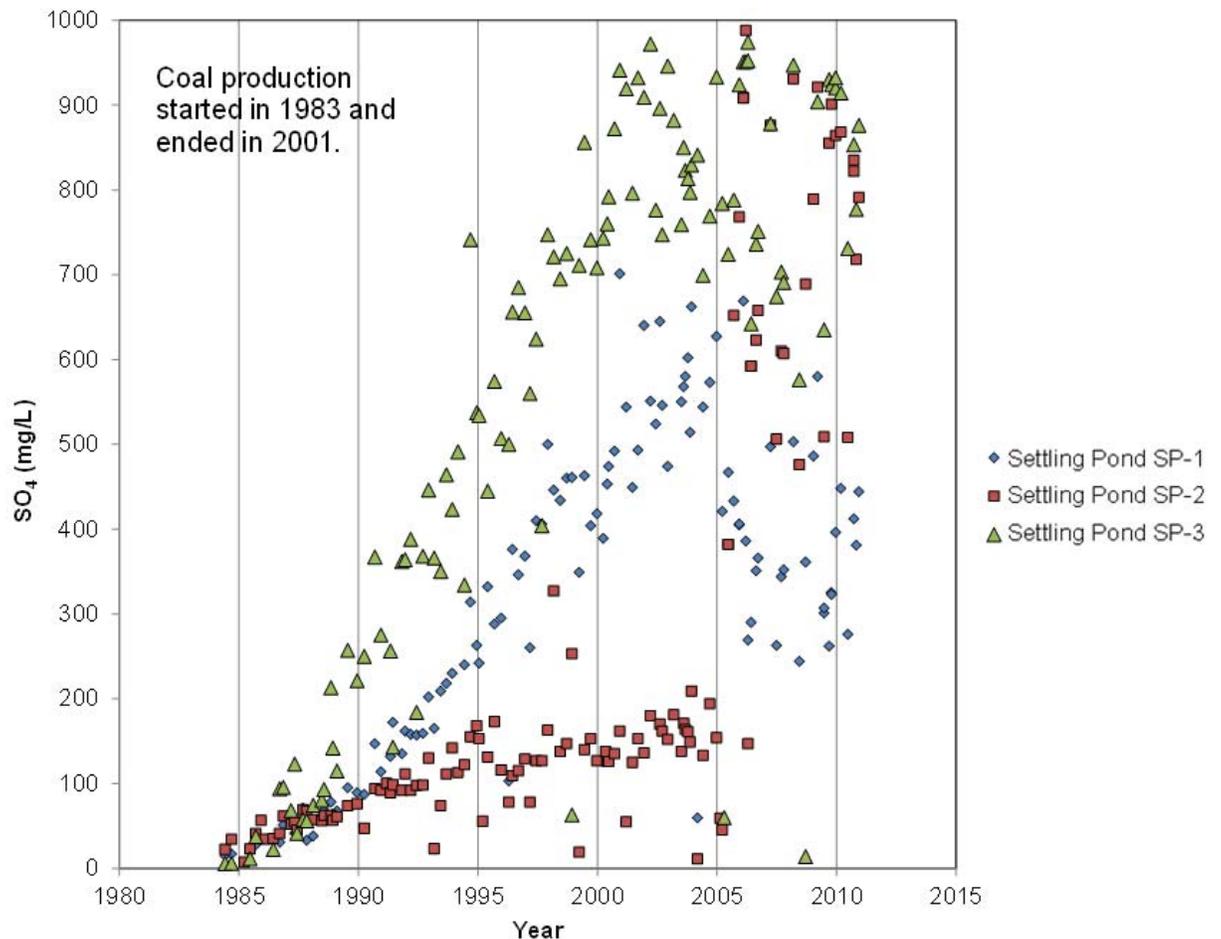
- **Oxidation of pyrite under oxygenated conditions:** This results in release of soluble components of pyrite, mainly sulphate and acidity, but also traces of elements including selenium and other metals. Release is followed by consumption of the resulting acidity by excess quantities of acid-neutralizing minerals and release of soluble components of those minerals, mainly base cations.
- **Interaction of trace elements with reactive surfaces, e.g. iron oxides:** Under dominantly basic conditions, this results in attenuation of elements in solution as cations, e.g. cadmium, cobalt, copper and zinc. Elements released as oxyanions, e.g. selenium and sulphate, remain mobile and show limited attenuation unless precipitated as secondary minerals, e.g., sulphate as gypsum and/or barite.
- **Leaching of explosives residuals contributes inorganic nitrogen (e.g., nitrate) to contact waters:** Since explosives are introduced during mining and nitrogen forms are not expected to be generated significantly by rock weathering, loadings of explosives residuals are expected to diminish with time.

In the context of the general approach described in Section 2.1, selenium, sulphate and nitrate are considered to be relatively soluble, because processes limiting their solubility are either not occurring or are poorly understood. As a result, leaching of these elements is calculated mainly based on rate rather than solubility, with exceptions as noted below. Based on the apparent regional correlation of cumulative waste rock volume with increasing selenium concentrations in the Elk River (Figure 1), the working assumption is that the rate of release of these parameters is a function of waste volume.

Leaching of elements forming cations in solution, including cadmium, calcium and magnesium, are calculated based on solubility. Load released is therefore a function of infiltration rate, which is generally proportional to facility footprint rather than volume.

The hydrological aspects of waste rock leaching are complex, and are being further evaluated to support overall water management in the Elk Valley. Empirically, leaching effects (both concentrations and loadings) are expected to vary seasonally in response to changes in infiltration caused by snowmelt and other climatological events. High flow events may expose more rock to leaching, resulting in higher chemical loads, but such events may also provide dilution, leading to lower concentrations.

The hydrological model also has implications for the potential for chemical loadings to lag behind waste placement, due to the need for wetting to occur before water can drain from the waste. These effects are also being investigated further (reference). For the purpose of this source term, lag in loadings is not explicitly considered, because monitoring data for the Elk Valley as a whole shows increasing selenium concentrations in the Elk River correlating with cumulative waste placement (Figure 1), with no clear lag effect. Monitoring data for from a Coal Mine in BC shows that sulphate concentrations began increasing in settling ponds shortly after the mine became fully operational in 1983 (Figure 3).



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Figure 3. Sulphate Concentrations in Settling Ponds at a Coal Mine in BC

2.5.2 Method Steps

The steps in the source-term method are described below. Derivation of inputs is provided in Section 3.

Step 1: Calculate Load Generated for the Soluble Elements Sulphate, Selenium and Nitrate

Based on the conceptual model used for this method, the average sulphate, selenium and nitrate loads generated and released are based primarily on the quantity of contributing waste rock, with less influence from the annual volume of infiltrating water, though there are seasonal and flow intensity effects as described subsequently. As a result, the method does not require well-defined infiltration estimates.

An initial estimate of the average annual selenium and sulphate load generated of each parameter j ($L_{i,j}$, in mg/year) was calculated as a function of average annual rate from:

$$L_{i,j} = V_{rock} \cdot R_j$$

where V_{rock} is the cumulative volume of rock at a disposal location in bank m^3 , and R_j is the generation rate for parameter j in $mg/m^3/year$.

Based on an observed relationship between normalized selenium release and normalized drainage flow (see Section 3.1.3), L_j can be calculated for any average annual flow condition using an empirical relationship, indicating that unitless flow and unitless release rates are nearly equivalent.

In contrast, nitrate is highly soluble and not generated by weathering of rock. R_j is expected to be high initially, and to decrease as the inventory of soluble nitrate decreases. The decay of nitrate may follow a non-zero-order relationship in which the amount leached is related to the amount remaining. The observed relationship is described in Section 3.1.3.

Step 2: Calculate Initial Concentrations

An initial estimate of average annual pore water concentration for parameter j ($C_{i,j}$, in mg/L) was calculated from:

$$C_{i,j} = \frac{L_{i,j}}{\bar{Q}}$$

where \bar{Q} is the average volume of water in L/year infiltrating through the waste.

Step 3: Evaluate Initial Sulphate and Selenium Concentrations for Potential Solubility Controls

Initial sulphate concentrations were evaluated against concentrations controlled by gypsum solubility. If the raw sulphate concentration exceeded the solubility of gypsum (C_{s,SO_4}), the initial concentration was reduced to C_{s,SO_4} . If the raw concentration was less than C_{s,SO_4} , the initial concentration was used.

Step 4: Add Concentrations of Less Soluble Parameters

Concentrations of other parameters ($C_{s,j}$) were then added to the source term at fixed levels or solubility limits.

Calcium and magnesium concentrations (C_{Ca} and C_{Mg} , in mg/L) were calculated to achieve an approximate ion balance using bicarbonate, sulphate and fluoride as the major anions:

$$C_{Ca} = \left(\frac{40}{2 \cdot (1 + M_{Mg/Ca})} \right) \cdot \left(\frac{2 \cdot C_{Alkalinity}}{100} + \frac{2 \cdot C_{SO_4}}{96} + \frac{C_{NO_3}}{14} + \frac{C_F}{19} - \frac{C_{Na}}{23} - \frac{C_K}{39} \right)$$

$$C_{Mg} = M_{Mg/Ca} \cdot \left(\frac{24}{40} \right) \cdot C_{Ca}$$

In these equations, the concentration in mg/L is divided by the molar mass (in mg/mmol; e.g., 96 mg/mmol for sulphate) and multiplied by the charge on the ion (e.g., -2 for sulphate). $M_{Mg/Ca}$ is the average molar ratio of magnesium to calcium in surface waters.

The resulting concentrations from Step 4 for each parameter j were calculated concentrations $C_{i,j}$.

Step 5: Calculate Load Released

Annual load released ($L_{a,j}$, in mg/year), for each parameter j was calculated from:

$$L_{a,j} = Q \cdot C_{c,j}$$

Where $C_{c,j}$, in mg/L is the pore water concentration for each parameter obtained from steps 3 and 4 and Q is the associated infiltrating flow.

Step 6: Calculate Monthly Load Distribution in Seepage

The monthly load distribution in seepage was calculated by distributing $L_{a,j}$ according to monthly fractions for each parameter $p_{m,j}$. The load released in month m ($L_{m,j}$, in mg/month) was:

$$L_{m,j} = L_{a,j} \cdot p_{m,j}$$

Step 7: Calculate Monthly Seepage Concentrations

Monthly concentrations ($C_{m,j}$, in mg/L), were calculated from the monthly flow distribution ($Q_{m,j}$, in L/month):

$$C_{m,j} = \frac{L_{m,j}}{Q_m}$$

Step 8: Compare Monthly Concentrations with Fixed Concentrations and Calculate Final Loadings

Values of $C_{m,j}$ were compared with constraining concentrations ($C_{s,j}$ in mg/L). If values of $C_{m,j}$ exceeded constraining values $C_{s,j}$, the constraining values replaced $C_{m,j}$.

The resulting final monthly concentrations ($C'_{m,j}$, in mg/L), can be used to calculate final loadings:

$$L'_{m,j} = Q_m \cdot C'_{m,j}$$

The total load released in a year ($L'_{a,j}$, in mg/year) is the sum of loads released in each month:

$$L'_{a,j} = \sum_{m=\text{January}}^{m=\text{December}} L'_{m,j}$$

Step 9: Speciate Nitrogen Forms

For the purpose of downstream predictions, nitrate concentrations were calculated and then used to speciate nitrate to nitrite and ammonia nitrogen according to the following:

$$C_{NO_2} \text{ (in mg N/L)} = f_{NO_2} \cdot C_{NO_3}$$

$$C_{NH_3} \text{ (in mg N/L)} = f_{NH_3} \cdot C_{NO_3}$$

where f_{NO_2} and f_{NH_3} are fixed factors used to calculate C_{NO_2} and C_{NH_3} from C_{NO_3} .

Step 10: Calculate Total Dissolved Solids

Total dissolved solids (TDS) concentrations (C_{TDS} , in mg/L), were estimated from the sum of all major ions as determined in Step 4:

$$C_{TDS} = C_{Ca} + C_{Mg} + C_K + C_{Na} + \frac{C_{Alkalinity}}{50} \cdot 61 + C_{SO_4} + C_{NO_3} + C_F + C_{Cl}$$

$\frac{C_{Alkalinity}}{50} \cdot 61$ was used to convert alkalinity concentrations, in mg CaCO₃/L, to bicarbonate concentrations, in mg/L.

2.6 Submerged Waste Rock

2.6.1 Conceptual Model

As waste rock is submerged (for example, when placed in a backfilled flooding pit), soluble weathering products not flushed by meteoric water may be released to the water column. For the purpose of the source term, the flushing process is assumed to be instantaneous, though in practice a rapid initial flush can be expected following by slow flushing of residual load.

Following this flushing process, the rock oxidizes at much lower rates than occurs under subaerial conditions due to low concentrations of dissolved oxygen in water relative to atmospheric conditions. Therefore, for the purpose of source term calculations, this load is considered to be zero based on extensive literature supporting subaqueous disposal as a technology to significantly limit sulphide mineral oxidation (The International Network on Acid Prevention 2009).

2.6.2 Calculation of Flushed Load

Stored load originates as weathering products in areas of the waste rock that have never been contacted by meteoric water, and in areas that are contacted, but in which weathering products are retained due to solubility limitations.

The one time load of sulphate, selenium and nitrate ($L_{\text{uncontacted},j}$, in mg) flushed from uncontacted parts of rock mass volume (V_{flooded} , in bank m^3) as they are inundated is:

$$L_{\text{uncontacted},j} = p_{\text{uncontacted}} \cdot V_{\text{flooded}} \cdot R_j \cdot (t_{\text{flood}} - t_{\text{placement}})$$

where $p_{\text{uncontacted}}$ is the proportion of rock not contacted by meteoric water, t_{flood} (year) is the time when flooding occurs and $t_{\text{placement}}$ (year) is the time when the rock was placed so that $(t_{\text{flood}} - t_{\text{placement}})$ is the time in years since exposure by mining.

The load flushed from contacted areas ($L_{\text{contacted},j}$ in mg) is:

$$L_{\text{contacted},j} = (1 - p_{\text{uncontacted}}) \cdot V_{\text{flooded}} \cdot R_j \cdot (t_{\text{flood}} - t_{\text{placement}}) - \sum_{a=t_{\text{placement}}}^{a=t_{\text{flood}}} L'_{a,j}$$

This expression is based on total load generated less the load removed by infiltrating water. For sulphate or selenium, this load may be zero if all concentrations are less than C_{s,SO_4} and $C_{s,Se}$, respectively.

Total load released by flushing ($L_{\text{flush},j}$, in mg) was calculated from:

$$L_{\text{flush},j} = L_{\text{uncontacted},j} + L_{\text{contacted},j}$$

2.7 Waste Rehandling

2.7.1 Conceptual Model

During rehandling, new flow paths are created within the waste during excavation resulting in leaching of previously unflushed waste by meteoric water. The flushed load reflects weathering products that have accumulated since the waste was originally deposited. Therefore, short-term increases in chemical loadings are expected, because the load from accumulated weathering products is added to ongoing load generation by and other weathering processes. These flushing processes are conceptually the same as those for Submerged Waste Rock (Section 2.6).

2.7.2 Calculation of Flushed Load

Flushed load was calculated using data obtained from water extraction tests (shake flask extraction, Price 2009) performed on legacy waste (waste rock, tailings, CCR, hot waste) samples from the Swift Project area at FRO. The load was calculated from the concentration in the extraction test multiplied by the volume of leachate divided by the volume of sample tested to yield load released on a one time basis when re-handled in mg/m³ of waste moved.

For legacy waste rock, the load released was reduced by a factor 0.2 to account for the difference in particle size used in the extraction test (-2 mm) compared to run of mine waste rock.

2.8 Pitwall Source Term

The method for pitwalls is the same as that for exposed waste rock, except that V_{rock} is calculated from the exposed area of the walls (A_{wall}) and an assumed reactive surface thickness (d):

$$V_{\text{wall}} = A_{\text{wall}} \cdot d$$

Exposed seam footwalls in pit floors tend to be composed of shales that may be more reactive than waste rock; however, the footwall is not blasted and is therefore not fractured to the same degree as walls in interburden rocks. Using V_{wall} to calculate leaching of the footwall is therefore considered to be conservative.

Subsequent calculations of $C'_{m,j}$ for wall rock runoff followed the same methods as described in Section 3.2.

2.9 Coal Rejects

2.9.1 Conceptual Model

Coal rejects may be coarse coal rejects (CCR) produced at EVO, GHO, and FRO, or coal refuse mixed with fines as produced at CMO and LCO. Coal rejects are typically placed in small dedicated facilities constructed in small lifts, and compacted.

Weathering processes in coal rejects are similar to waste rock; however, monitoring of gas concentrations in the Greenhills Area A CCR dump has shown that oxygen penetration into coal reject dumps may be limited by oxygen-consuming reactions (SRK 2013b). As a result, leaching was assumed to be a function of the footprint area of the coal rejects facilities, rather than their volume, with oxygen penetration occurring to a fixed depth.

In addition, the presence of coal fines in coal rejects indicates that reactive surfaces may serve to control trace element concentrations to low levels, and the presence of oxygen-limited conditions may limit Se leaching by transformation to chemically reduced forms.

If CCR is co-disposed with waste rock, it may oxidize without oxygen limitation, like the associated waste rock.

2.9.2 Calculation of Source Term

CCR Dumps

The source term for CCR consists of fixed concentrations applied to all infiltrating waters, based on the conceptual model and empirical data (SRK 2012a, 2013b). As a result, loadings are a function of assumed infiltration.

CCR Co-Disposed with Waste Rock

The source term for co-disposed waste rock and CCR is calculated from the sum of waste rock rates and CCR weathering rates under fully atmospheric conditions. No site drainage data are available from which to calculate release rates for co-disposed CCR under site conditions. Co-disposed CCR release rates are therefore based on laboratory rates indicated by humidity cells (for example, SRK 2013b and unpublished data).

Laboratory rates were decreased by a factor to account for the lower temperatures onsite, and decreased by a conventional generic factor of 50% to allow for incomplete contact with infiltrating water. This is near the high end of the range reported by Kempton (2012).

3 Input Data

3.1 Derivation of R_j and Related Factors for Waste Rock Using Analog Datasets

Seasonal distribution of loadings, R_j , (j =selenium, sulphate and nitrate) and relationships between annual load release and flow were developed by interpretation of monitoring datasets provided by Teck.

3.1.1 Database Description

Three datasets were needed to calculate inputs into the source-term method for waste rock:

- Near-source water quality data for monitoring points downstream of waste rock dumps.
- Accompanying flow data for the same locations.
- Waste rock quantities upstream of the monitoring point.

Teck provided the data for each component. The need for all three components was assessed and used as the basis to determine which sites could be used for the calculation. Figure 4 shows the availability of data by operation and monitoring location.

Fording River Operation (FRO)

Teck provided waste rock quantities for ten monitoring points at FRO. Based on review of the available water quality and flow data, two of the site locations (Kilmarnock Creek [KC] and Henretta Creek [HC]) had the required coincident waste rock volumes, water chemistry and flow data.

Kilmarnock Creek drains mainly the Brownie and Kilmarnock spoils, as well as a large undisturbed catchment to the south of the current FRO mining area. The catchment contains over 1 billion m^3 of spoil placed since 1980. The weighted average age of the spoil was 11 years calculated from:

$$\text{WeightedAge(years)} = 2011 - \frac{\sum_{t=2011}^{t=\text{start}} V_t \cdot t}{\sum_{t=2011}^{t=\text{start}} V_t}$$

where V_t is the volume of rock placed in year t . Year 2011 was the latest year for which waste rock quantities were available.

Henretta Creek mainly drains spoil from Henretta Ridge. Mining on the ridge began in 1992 and is continuing. The weighted average age of the spoil was 10 years in 2011. The quantity of spoils to the end of 2011 was approximately 150 million m^3 .

Greenhills Operation (GHO)

Teck provided waste rock quantities for seven drainage basins on Greenhills Ridge (Swift, Cataract, Porter, Greenhills, Thompson, Wolfram and Leask). Chemistry and flow data were available for three monitoring locations (Cataract, Swift, Porter). All three creeks drain catchments are dominated by waste rock, with relatively low proportions of undisturbed areas.

Cataract Creek (CC) drains approximately 400 million bank cubic metre (BCM) of waste rock placed since 1984. The average age of the spoil was 13 years.

Swift Creek drains approximately 213 million BCM of waste rock placed between 1993 and 2006. The average age of the spoil was 14 years.

Porter Creek (PC) drains approximately 79 million BCM of waste rock placed since 1986. The average age of the spoil was 16 years.

Line Creek Operations (LCO)

LCO tracks waste rock placement at 13 locations. Drainage monitoring occurs in West Line Creek and Line Creek. Two suitable monitoring locations were West Line Creek (WLC), which drains the WLC spoil, and Line Creek (LC) downstream of LCO (LC3).

Flow data for WLC were available for 2010 and 2011. Due to the more extensive database available, LC3 was also included, though it represents the entire drainage from the site including WLC.

Elkview Operations (EVO)

EVO tracks waste placement in seven surface water catchments (Bodie Creek, Dry Creek, Erickson Creek, Goddard Creek, Harmer Creek, Michel Creek and Six Mile Creek). Except for Goddard and Michel creeks, water quality monitoring data from all locations could be used for this evaluation. Michel Creek was unsuitable because it is in a large catchment that has other influences including CMO.

Of the available sites at EVO, suitable datasets were available for Bodie (BC1), Harmer (HC1), Erickson (EC1) and Six Mile Creeks (SM1). Gate Creek (GT1) also had a suitable monitoring database but no assigned waste rock quantity. Bodie Creek's catchment was dominated by spoil. Harmer, Erickson and Six Mile Creeks had spoils in their headwaters but also significant proportions of their catchments undisturbed by mining activities.

Bodie Creek contained a total spoil volume of approximately 243 million BCM placed since 1993, with an average age of eight years in 2010, the latest year for which spoil volumes were available. Bodie Creek is highly influenced by pit dewatering.

The flow and water quality monitoring point in Harmer Creek (HC1) reflected loadings contributed from spoils assigned to Harmer Creek and Dry Creek. The total spoil volume was approximately 557 million BCM with an average age of 23 years. Spoil had been placed there since 1969, with a brief hiatus between 1999 and 2003.

Erickson Creek (EC1) contains a spoil volume of approximately 399 million BCM placed continuously since 1969, with an average age of 18 years.

Six Mile Creek contains approximately 7 million BCM of spoil, all placed in 1980.

Coal Mountain Operations (CMO)

CMO tracks spoil placement at five locations, two of which report to the Corbin Creek Pond monitoring location which has suitable monitoring data from 2004. Total volume of spoil placed since 1985 in 2009 was 157 million BCM, with an average age of nine years in 2011.

3.1.2 Data Processing

Steps used to calculate selenium, sulphate and nitrate loading rates from the monitoring data are described below. The same procedure was used for each site.

Step A – Calculate Daily Loads

Monitoring data were available on up to a daily basis for flows, and much less frequently for water chemistry. In concept, this information permitted simulation of a continuous record of daily loadings.

Missing concentration and flow values for individual days were extrapolated by using the last date for which daily flow and chemistry data were recorded. For example, if flow data were collected on May 10 and May 13, flows for May 11 and 12 were based on the May 10 measurement. The same approach was used for concentrations. The combined flow (Q) and concentration data (C_j) for parameter j on each dataset were used to calculate loadings (L_{j,day}) for parameter j using the conventional formula:

$$L_{j,day} \text{ (mg/day)} = Q \text{ (m}^3\text{/day)} \cdot 1000 \text{ (L/m}^3\text{)} \cdot C_j \text{ (mg/L)}$$

Step B – Calculate Annual Loads

Annual loads (L_{j,year}, in mg/year) were then calculated by summing the daily loads for each parameter.

$$L_{i,year} = \sum_{day=January 1, Year}^{day=December 31, Year} L_{i,day}$$

Only years for which there were at least six monitoring rounds for flow and chemistry were included. This target was based mainly on having data for at least six months representing the six months of open water (May to October) to ensure that highest chemical loadings during the snowmelt period were included in the assessment.

Step C – Calculate Annual Load Leached on Rock Mass Basis

The load leached per volume of rock ($L'_{i,year}$, in mg/year/m³) was calculated based on the cumulative volume of rock in a year indicated by the waste rock placement history:

$$L'_{year} = \frac{L_{i,year}}{\sum_{t=start}^{t=year} V_t}$$

Step D – Calculate Average Annual Loads

Loading rates obtained in Step B for each site were used to calculate average annual loads.

Step E – Calculation of Monthly Loading Distribution

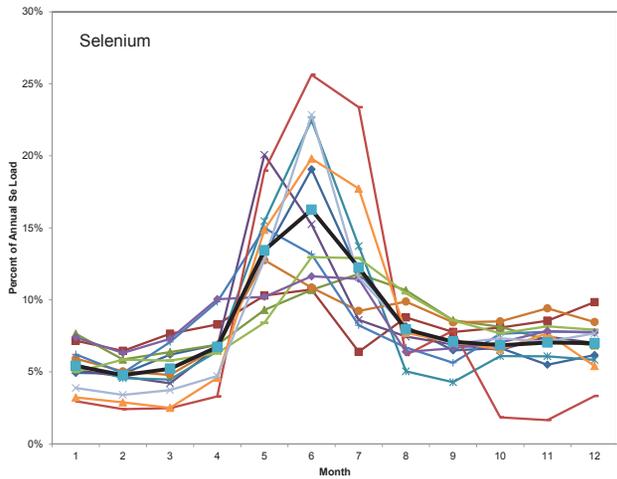
Loadings on monthly time-step was calculated for years in which data were available, by summing daily loads in each month. The proportion of load in each month is therefore the load released in the month divided by the annual load. These distributions were developed using data from existing spoils that are either active or have been reclaimed using conventional revegetation techniques. Different distributions can be expected when spoils have been covered with materials intended to significantly modify infiltration.

3.1.3 Results

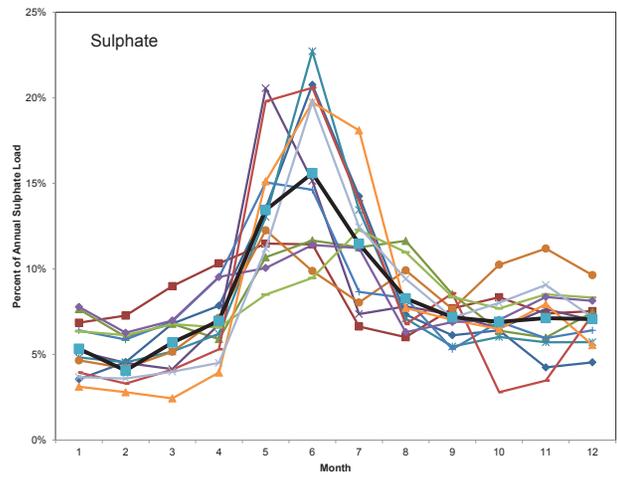
Description of Trends

Monthly distributions of the fraction of flow, and selenium, sulphate and nitrate loadings are shown in Figure 5. All sites experienced peak flows typically between late March and May. Erickson Creek appeared to have a late flow peak in June and August, but flow information was limited and Erickson Creek is sub-surface in some sections.

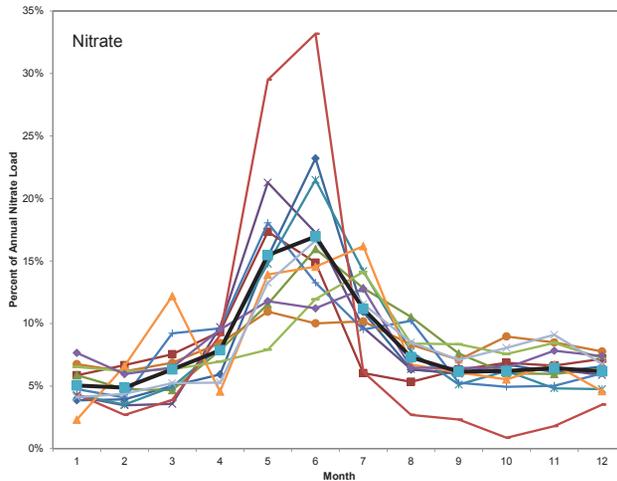
The relationship between flow peak and loadings showed some consistency. Greatest loadings were commonly apparent during the flow peak. The strong similarity of loading and flow trends imply that seasonal flushing of soluble components occurs in response to increased infiltration during the melting of snow.



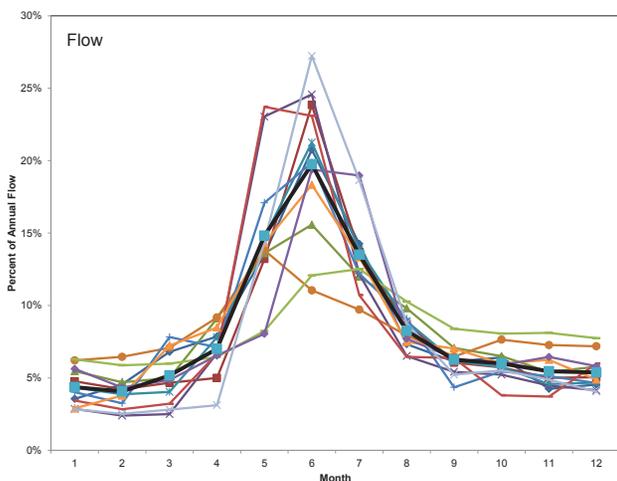
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Figure 5. Monthly Distribution of Load and Flows

Trends in Annual Loadings

Trends in annual loadings were evaluated when data were available as total loadings (L in mg/year or g/year), and normalized with respect to cumulative waste rock volume.

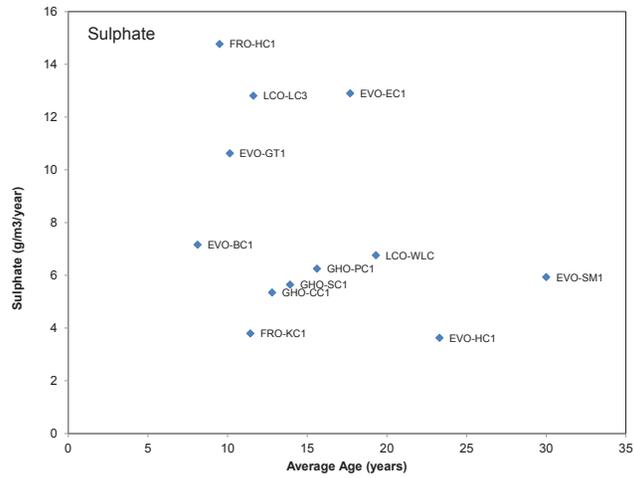
There was a lack of consistent trends for both expressions of load for sulphate and selenium, and a lack of sufficient data points to evaluate trends. Nitrate total load either showed no trend, or else increasing trends that discontinued or decreased when normalized to waste rock volume.

Monitoring data showed a relationship between average age (\bar{t} in years) of waste rock and leaching nitrate (Figure 6). Equations of the general form could be fitted to the trend:

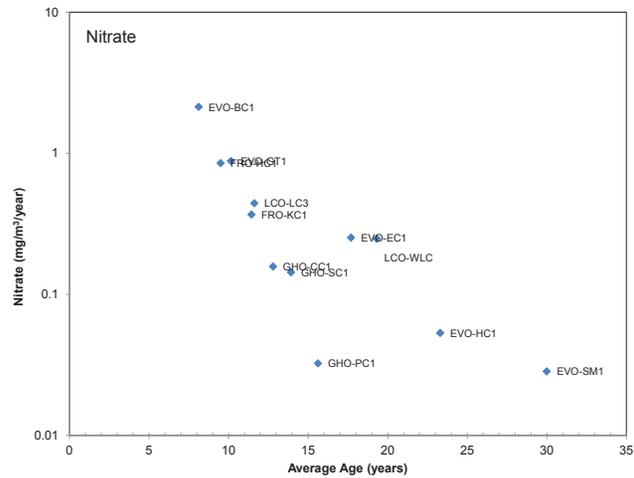
$$L_{i,NO_3} = 10^{-a_k \log \bar{t} + a_1}$$

No evidence of a similar relationship was apparent for selenium or sulphate (Figure 6).

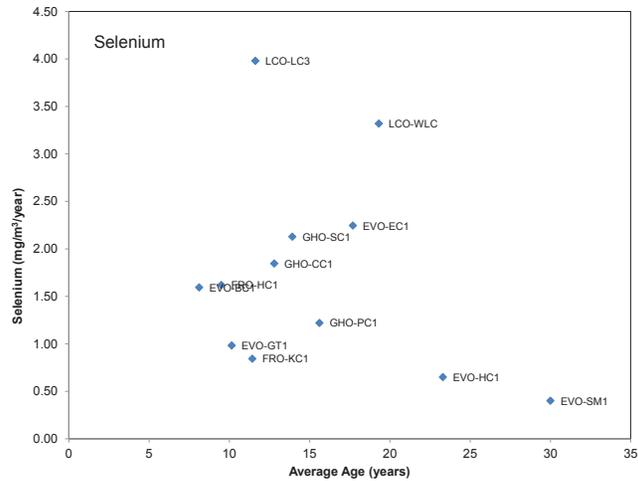
Correlation coefficients for selenium and sulphate were not significantly different from zero at a significance level of 0.05; and therefore do not allow change in release rates as a function of spoil age to be incorporated into the method. Coefficients for the nitrate equations are provided in Table 3.



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Figure 6. Relationship Between Age of Spoil and Average Release Rates

Average Normalized Load for Selenium and Sulphate

Average waste volume normalized loads for each site were calculated using individual annual average measurements. An indication of the spread of data was obtained by calculating the 95% confidence limits on the mean (Figure 7). The range of normalized loads for each parameter is narrow and nearly within an order of magnitude in each case.

Average selenium and sulphate release rates and ranges are provided in Table 3 for operations other than CMO, which are provided in Table 4. Nitrate release rates are the same for all sites.

Table 3. Generation Rates (R_j) – Operations Other Than CMO

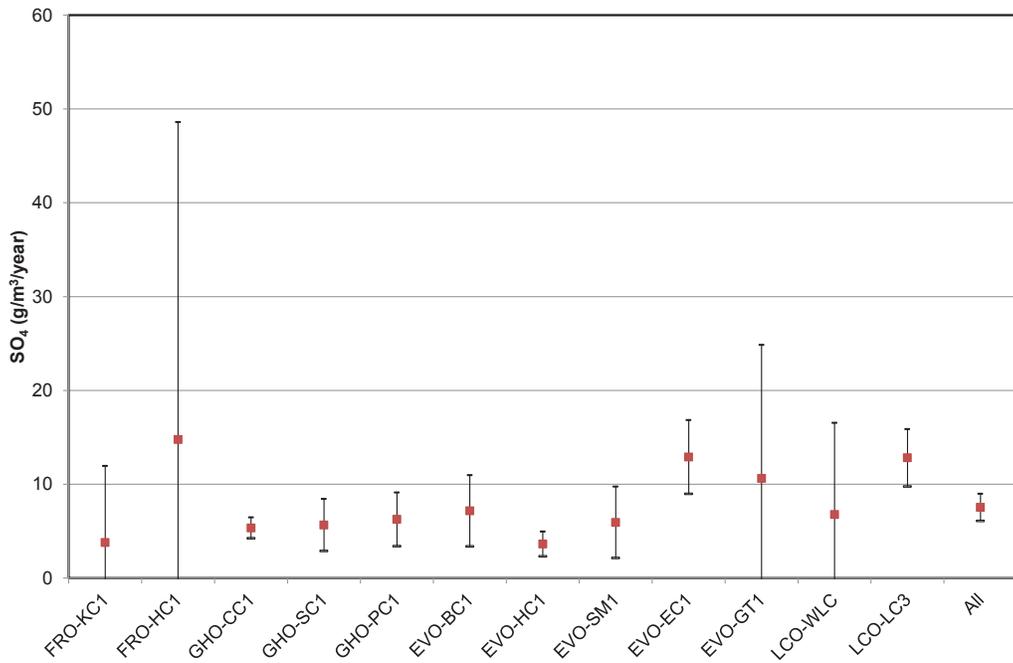
Case	Based on Monitoring Data to 2011		
	SO ₄	Se	NO ₃
	n=69	n=77	n=128
	g/m ³ /year	mg/m ³ /year	g N/m ³ /year
Arithmetic Average	7.5	1.6	10 ^{-2.9log\bar{t}+2.7}
95% Upper Confidence Limit	9	1.9	10 ^{-3.0log\bar{t}+3.0}

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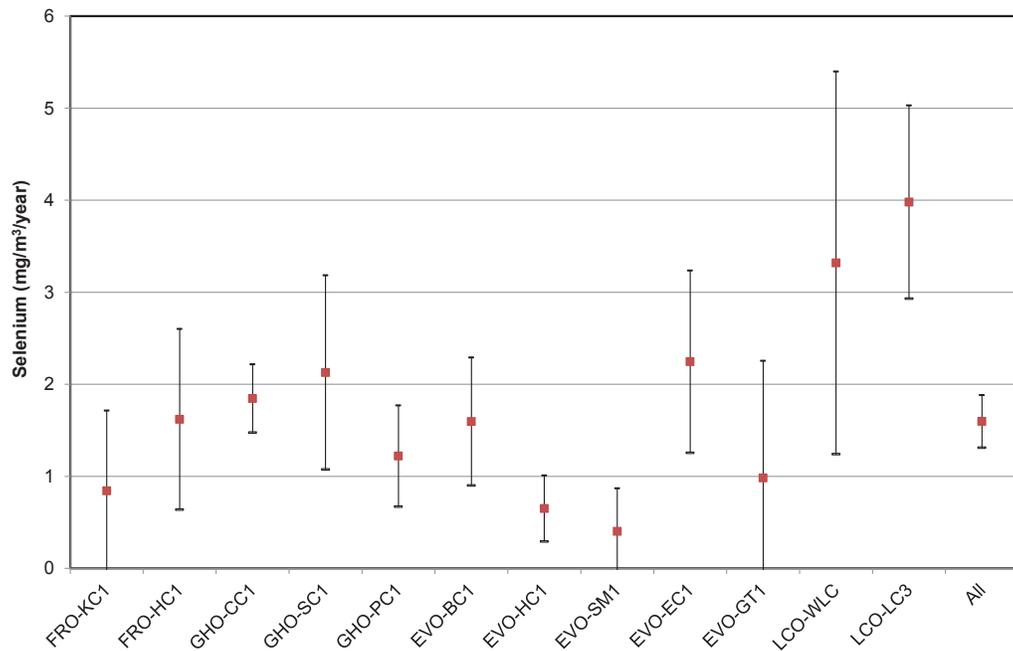
Table 4. Summary of CMO Generation Rates

Case	Based on Monitoring Data to 2011		
	SO ₄	Se	NO ₃
	n=5	n=5	n=128
	g/m ³ /year	mg/m ³ /year	g N/m ³ /year
Arithmetic Average	17	0.55	10 ^{-2.9log\bar{t}+2.7}
95% Upper Confidence Limit	27	1.0	10 ^{-3.0log\bar{t}+3.0}

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Figure 7. Average Annual Loads and Confidence Limits for Sulphate and Selenium

Seasonal Variations

Average monthly load distributions obtained from the data shown in Figure 3 are provided in Table 5.

Table 5. Average Monthly Loading Distribution (p_{m,i})

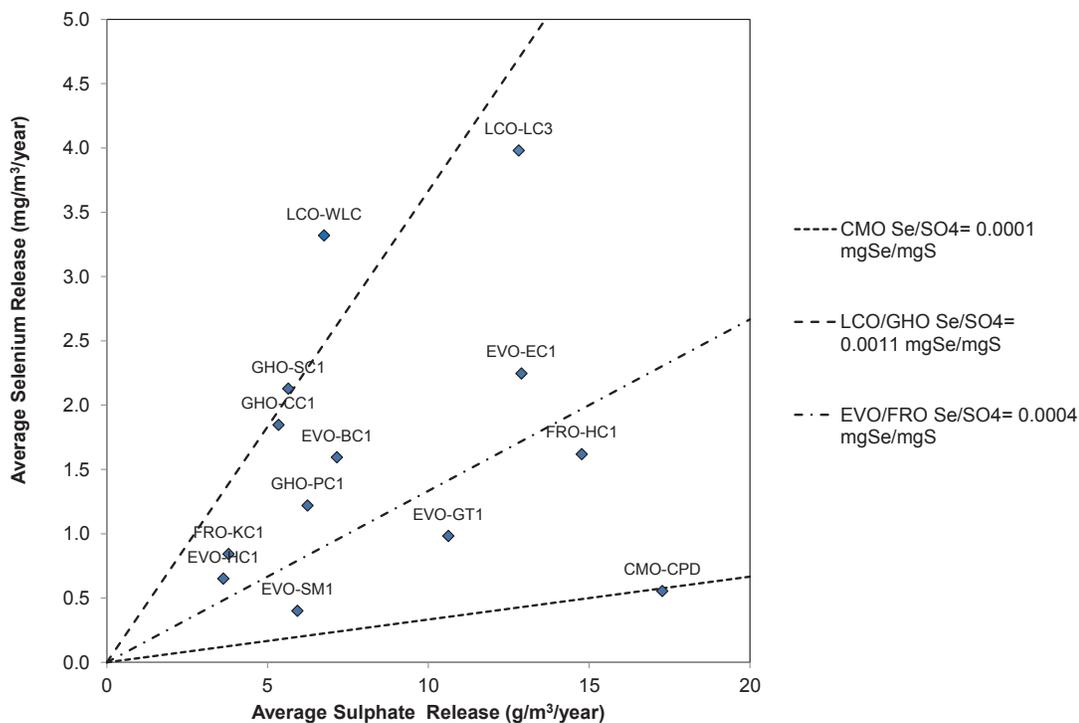
Month	Sulphate	Selenium	Nitrate
January	4%	5%	5%
February	4%	5%	5%
March	5%	5%	6%
April	7%	7%	8%
May	15%	13%	15%
June	20%	16%	17%
July	14%	12%	11%
August	8%	8%	7%
September	6%	7%	6%
October	6%	7%	6%
November	5%	7%	6%
December	5%	7%	6%

Source: \\van-svr01\Projects\02_MULTI_SITES\Elk_Valley_Coal_Corp\1CE003.001_Selenium_Geochemistry\Selenium_Release_Model\2012-06_Loading_Empirical_Report_2012_Update\LoadingCalculations_1CE003001_REV12.xlsx

Comparison of Sulphate and Selenium Release

The conceptual model indicates sulphate and selenium release ought to be correlated because they are released by the same process (pyrite oxidation). Figure 8 compares average sulphate and selenium release rates. There is weak evidence of a regional correlation between sulphate and selenium; however, some site differences are apparent when considering the ratio of selenium to sulphate

- The ratio of selenium to sulphate is greatest at 0.0011 mgSe/mgS for LCO WLC and two monitoring locations at GHO (SC1 and CC1). GHO_PC1 (Porter Creek) has a lower ratio, though the drainage from that location may be influenced by saturated fills in historical pits, which causes the ratio to decrease due to Se attenuation (SRK 2013a). LCO_LC3 also shows a high ratio, but this monitoring location is influenced by LCO_WLC.
- At 0.0001 mgSe/mgS, the ratio at CMO is an order of magnitude lower than LCO and GHO.
- EVO and FRO sites show an intermediate ratio near 0.0004 mgSe/mgS. This is four times the CMO ratio and about one third the ratio for LCO and GHO. EVO-SM1 showed the lowest ratio in this group, which was also the oldest average spoil in the Elk Valley.



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Figure 8. Comparison of Average Sulphate and Selenium Release Rates for Arithmetic Averages

The relationships between release of nitrate, and release of sulphate, and release of nitrate and release of selenium were investigated, but none were apparent, which is consistent with the conceptual model.

Relationship between Flow and Selenium Loading

Figure 9 shows annual flow and selenium loading as ratios to the averages for each monitoring location. A strong positive correlation indicates that as flow increases, annual selenium load also increases. The regression relationship is statistically the same as the equivalence line along which average normalized flow is equal to average normalized load. As a result, there does not appear to be a tendency for higher loads to be disproportionately associated with higher flows, though there is a tendency for scatter in the data points shown in Figure 9 to increase at higher loads and flows.

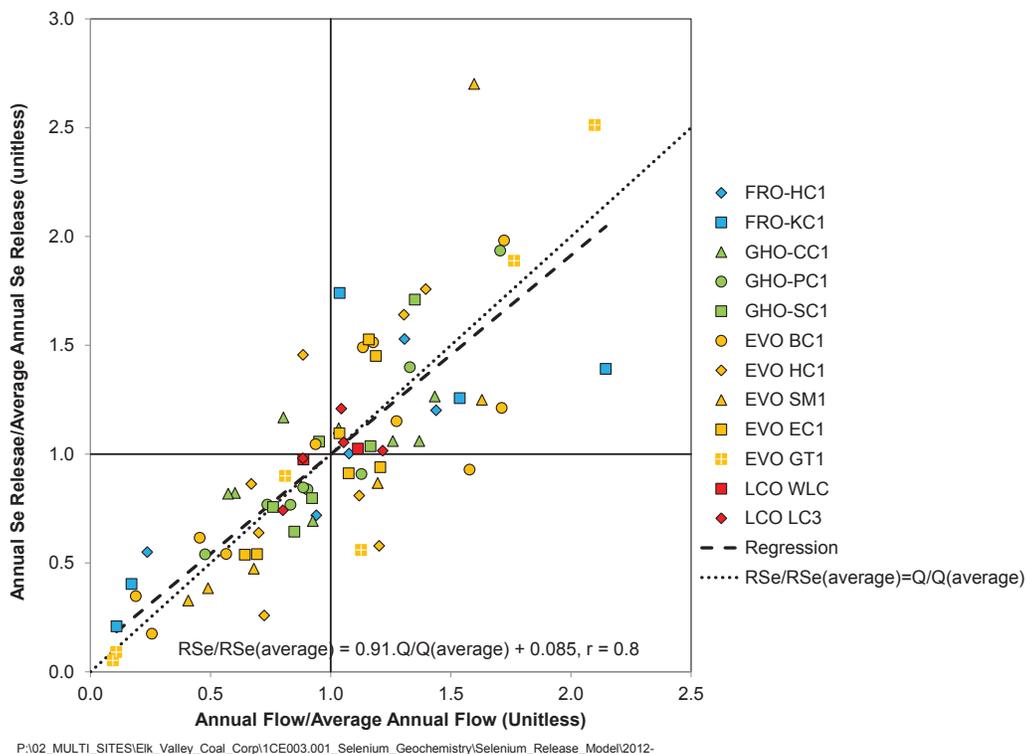
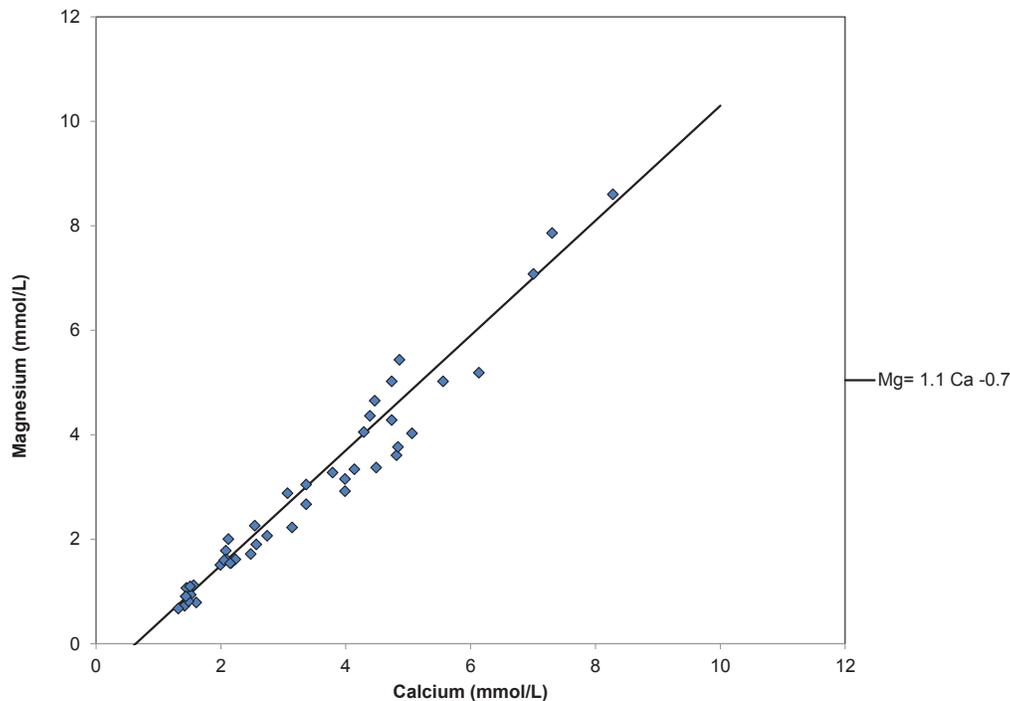


Figure 9. Relationship between Normalized Annual Flow and Normalized Annual Selenium Loading

3.1.4 Element Solubility Constraints

Calcium to Magnesium Molar Ratio ($M_{Mg/Ca}$)

The value of 1.1 for $M_{Mg/Ca}$ was calculated based on the slope of the relationship observed between Ca and Mg in drainage data throughout the Elk Valley available in 2010 (Figure 10).



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Figure 10. Relationship of Ca and Mg in Elk Valley Spoil Drainage Used to Estimate $M_{Mg/Ca}$

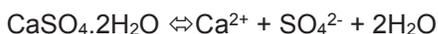
Sulphate

Solubility limits for individual dissolved ions are often a result of the finite solubility of minerals containing the ions. Gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) is a mineral commonly considered to exert strong control on sulphate solubility because it is observed as a product of rock weathering and can form at typical mine drainage sulphate concentrations. Other sulphate minerals that form from common major ions, including magnesium, potassium and sodium, are much more soluble.

The relationship defining the solubility of gypsum and the resulting concentration in the solution is the solubility product¹:

$$k_{sp, \text{gypsum}} = [\text{Ca}^{2+}] \cdot [\text{SO}_4^{2-}]$$

$[\text{Ca}^{2+}]$ and $[\text{SO}_4^{2-}]$ are the concentrations of calcium and sulphate in water, and k_{sp} is the equilibrium constant for the dissolution of gypsum in water, which is defined by the thermodynamics of the reaction:



¹ Concentration is strictly expressed as activity, which is the concentration adjusted by the activity coefficient (γ). For simplicity in this report, $\gamma=1$ for all ions.

The solubility product indicates the range of calcium and sulphate concentrations in solution that are consistent with the presence of the mineral gypsum. If gypsum dissolves in pure water, the resulting calcium and sulphate concentrations will be exactly equal when expressed in moles. However, if sulphate and/or calcium originate from other sources, sulphate concentrations for water in equilibrium with gypsum are defined by:

$$[\text{SO}_4^{2-}] = \frac{k_{\text{sp, gypsum}}}{[\text{Ca}^{2+}]}$$

The solution must remain in charge balance, so that as sulphate increases relative to calcium, other positive ions must be present. For example, if magnesium is in solution, the charge balance is:

$$[\text{SO}_4^{2-}] = [\text{Ca}^{2+}] + [\text{Mg}^{2+}]$$

If gypsum is present but a magnesium sulphate (e.g., epsomite $[\text{MgSO}_4 \cdot 7\text{H}_2\text{O}]$) is not, the gypsum solubility product and charge balance equations can be solved to calculate the concentration of sulphate as a function of magnesium concentration and $k_{\text{sp, gypsum}}$:

$$[\text{SO}_4^{2-}] = \frac{[\text{Mg}^{2+}] + \sqrt{[\text{Mg}^{2+}]^2 + 4k_{\text{sp, gypsum}}}}{2}$$

The equation indicates that the concentration of sulphate in equilibrium with gypsum increases as the magnesium concentration increases.

Calculation of Sulphate Solubility for the Elk Valley

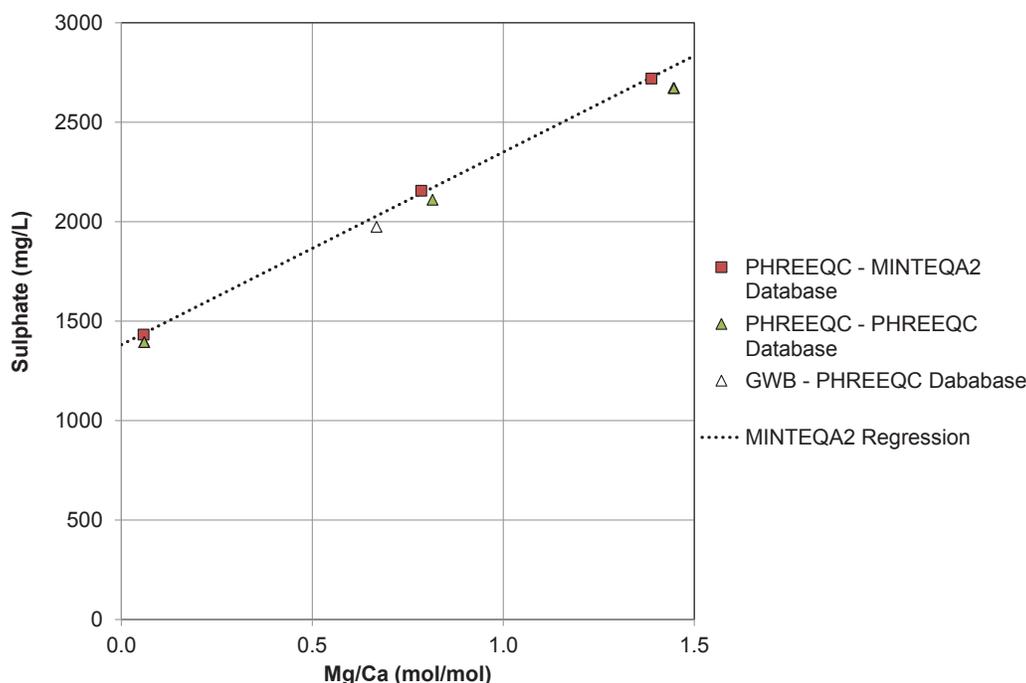
For the purpose of calculating sulphate solubility in the current source-term method, the following assumptions were made:

- Gypsum is a logical controlling secondary mineral due to the presence of abundant calcium-containing carbonate in the Mist Mountain Formation.
- The presence of dolomite ($\text{CaMg}(\text{CO}_3)_2$) in the host rocks is important because it contributes calcium and magnesium in equal molar proportions and therefore can be expected to elevate the gypsum solubility limit above that expected for calcium only.
- Data collected as part of the Applied R&D Program shows that waste rock pore waters may be at equilibrium with gypsum. Therefore, the Mg/Ca ratio in drainage waters can be used to calculate the sulphate concentration in equilibrium with gypsum.

Sulphate concentrations for waters in equilibrium with gypsum as a function of the Mg/Ca ratio were calculated using the public-domain equilibrium thermodynamic modelling software PHREEQC. Main inputs were thermodynamic data that describe the solubility of minerals and the speciation of ions in solution.

Since the thermodynamic databases are experimentally-derived and can be variable, the two publicly-available databases of Allison et al. (1991, MINTEQA2, United States Environmental Protection Agency) and Parkhurst and Appelo (1999, PHREEQC, United States Geological Survey) were used as inputs. In addition, since algorithms used to perform the calculations can be different, Geochemist's Workbench (GWB) software (Bethke 2009) was used as a check of the calculation method using the PHREEQC database.

Figure 11 shows sulphate concentrations in equilibrium with gypsum as a function of Mg/Ca at a temperature of 5°C. The individual plots indicate model predictions that are highly correlated with Mg/Ca. The two databases yielded nearly the same relationships, and use of the GWB software did not affect the solubility relationship calculated using PHREEQC. The MINTEQA2 database yielded slightly higher concentrations.



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Figure 11. Modelled Sulphate Concentrations as a Function of Mg/Ca Ratio Using Two Databases and Two Geochemical Modelling Programs

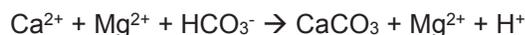
Figure 10 shows the strong correlation between calcium and magnesium for drainage waters from spoils in the Elk Valley. The slope of the regression line is Mg/Ca=1.1, which can be used to calculate a gypsum-constrained sulphate concentration of 2400 mg/L. For comparison, sulphate concentration in equilibrium with pure gypsum in the absence of magnesium (i.e., Mg/Ca = 0) is calculated to be 1400 mg/L.

The observed Mg/Ca ratio (i.e., the slope of the regression equation) is greater than the most Mg-enriched mineral (dolomite, $\text{CaMg}(\text{CO}_3)_2$) in the rock, implying that leaching processes are leading to preferential enrichment of Mg in drainage waters relative to Ca. The following linked processes may explain the ratio above 1.

Reaction of acidity generated by sulphide oxidation with dolomite yields water with molar equivalent Ca and Mg concentrations:



As Ca^{2+} and HCO_3^- concentrations increase, calcite saturation will be reached, causing Ca^{2+} to be removed from solution as calcite (CaCO_3):



However, dolomite or magnesite (MgCO_3) are not expected to precipitate in the same way, due to their slow kinetics of formation at atmospheric temperatures. A number of factors preclude unlimited enrichment of magnesium in waters, including:

- The presence of primary calcite (CaCO_3) and ankerite ($\text{CaFe}(\text{CO}_3)_2$) in the rock, which adds Ca^{2+} to water without adding Mg^{2+}
- The role of H^+ (i.e., pH), which causes an equilibrium to be reached
- The decreasing tendency for calcite to form and remove Ca as the Mg/Ca ratio in the water increases
- Co-precipitation of Mg in calcite.

Together, this appears to provide the explanation for the regional consistency of Mg/Ca in Elk Valley drainage waters shown in Figure 10. As a result, sulphate concentrations can be constrained using the relationship shown in Figure 11.

Discussion of Evidence for Sulphate Solubility Limits

Drainage Chemistry in the Elk Valley

Recent data for concentrations of sulphate in mine drainages in the Elk Valley show that the highest concentrations measured approach 1800 mg/L in the drainages from GHO. As of 2012 at Swift Creek and Cataract Creek, the sulphate concentration trend currently appears to be upward, though a significant decrease in the rate of increase occurred in 2010 and has continued. The well-studied West Line Creek Dump at LCO has yielded a maximum sulphate concentration of 1300 mg/L on a stable trend. The maximum sulphate concentration in drainage from GHO's Area A CCR dump is 1900 mg/L.

These monitoring locations are all affected to some degree by dilution with non-contact waters and the mixing of different types of contact waters. An estimate of the dilution fraction can be obtained from the disturbed vs. non-disturbed areas for monitoring location catchments, though it

is more complicated in reality. At West Line Creek, about one-third of the catchment area has spoil, which would lead to actual maximum concentrations of 3,900 mg/L from the spoil. However, maximum concentrations are measured in late winter when the contribution from the spoil is probably much greater than from one-third of the catchment area. As a result, maximum pore water concentrations will be less than 3900 mg/L as shown below.

Chemistry of Pore Waters in West Line Creek Dump

Pore water chemistry is being determined as part of the Applied R&D Program work on selenium by squeezing waste rock samples. At West Line Creek, maximum pore water concentrations of about 2000 mg/L have been measured after excluding one anomalous value above this concentration. These pore water concentrations exceed concentrations in the overall West Line Creek drainage, suggesting that West Line Creek is influenced by non-contact dilution and that a solubility control at or exceeding 2,000 mg/L may be supported.

Mineralogical Evidence for Sulphate Minerals in the Elk Valley

SRK (2004) documented the presence of gypsum in weathered pitwalls at EVO though it is not known if this is primary or secondary gypsum.

Mineralogical work by the University of Saskatchewan on the West Line Creek Dump samples has not detected the presence of gypsum, though barium sulphate (BaSO_4 , barite) has been identified microscopically. SRK (2004) also identified barite at EVO.

Barite is currently being evaluated by the University of Saskatchewan for its role in sulphate solubility control and in sequestering selenium. SRK considers that it is unlikely that barite will be a significant sulphate control because the availability of barium is limited. While barite has a much lower solubility than gypsum, barium is released by weathering of slow-reacting minerals such as feldspars, so that the presence of calcium and magnesium from carbonate mineral dissolution will drive sulphate solubility.

Selenium

A solubility limit for selenium was based on the assumption selenium can co-precipitate with gypsum (Fernández-González et al. 2006). A $C_{s,Se}$ solubility limit concentration of 1.5 mg/L selenium was estimated using the sulphate concentration of 2400 mg/L and a typical Se/SO_4 ratio. The range of Se/SO_4 ratios observed in the valley indicates that this concentration would vary by site.

There is ongoing work evaluating selenium solubility controls, including contact-water selenium concentrations and selenium co-precipitation with sulphate minerals such as gypsum and barite (reference).

No solubility controls were used for nitrate, due to its high solubility relative to concentrations in the Elk Valley.

From monitoring data, the speciation of nitrogen forms (f_{NO_2} , f_{NH_3}) as a function of nitrate was estimated as:

$$C_{NO_2} \text{ (mg N/L)} = 0.0063 \cdot C_{NO_3}$$

$$C_{NH_3} \text{ (mg N/L)} = 0.012 \cdot C_{NO_3}$$

Fixed solubilities of other elements used to calculate overall water chemistry are shown in Table 6. These statistics were calculated based on a consolidated database of water chemistry for operations in the Elk Valley, including field weathering leach-pile experiments at LCO (SRK 2013b).

The resulting chemistry shown in Table 6 is approximately in equilibrium with calcite, giving a partial carbon dioxide pressure of 10%.

The waste rock contact factor ($p_{uncontacted}$) for use in the Submerged Waste Rock Source Term (Section 2.6.1) was assigned a value of 0.5, i.e., 50% of rock is not flushed by meteoric water.

Table 6. Waste Rock Source-Term Element Concentrations

Parameter Type	Element	Units	Based on Monitoring Data to June 2010	
			P ₅₀	P ₉₅
Concentrations used to adjust $C_{i,j}$	SO ₄	mg/L	2400	2400
	Se	mg/L	1.5	1.5
Parameters added to source term at fixed concentration	pH	s.u.	8.2	8.4
	Alkalinity	mgCaCO ₃ /L	330	373
	F	mg/L	0.66	1.2
	Cd	mg/L	0.0011	0.0029
	K	mg/L	2.6	6.5
	Na	mg/L	8.4	15

Source: P:\02_MULTI_SITES\Elk_Valley_Coal_Corp\1CE003.001_Selenium_Geochemistry\Selenium_Release_Model\2012-06_Loading_Empirical_Report\Other_Element_Scaling\1CE003.001_Compiled_Results_SJD_Id_20110221_VER02.xlsx]

3.2 Re-Handled Wastes

Load released by re-handling of wastes are provided in Table 7. The 50th and 95th percentiles of shake flask leachate concentrations were used to represent the range in loads for each type of legacy waste. For any parameters not shown in Table 7, load released was determined from the volume of contact water and fixed concentrations shown in Table 6 (legacy waste rock and hot waste) and Table 8 (legacy CCR and tailings).

3.3 Pitwall

The thickness depth (d) was set at 2 m, which is a typical overblast depth for mining in the Elk Valley.

Table 7. Load Released by Re-Handling of Legacy Wastes (mg/m³)

Parameter	Historic Tailings		Historic CCR		Historic Waste Rock		Historic Hot waste	
	P50	P95	P50	P95	P50	P95	P50	P95
Ca	85000	130000	33000	65000	16000	23000	220000	1200000
Cd	0.12	0.25	0.4	1.2	0.043	0.11	0.26	5.2
Cl	2000	4400	3400	6000	510	780	5100	7300
F	1700	2300	820	1900	350	550	3100	3500
Mg	34000	83000	11000	26000	6300	8500	100000	400000
Na	1500	4100	840	1200	480	1100	1400	58000
Nitrate (N)	230	600	4500	13000	1500	2700	1000	13000
Se	77	89	22	34	15	31	30	70
SO ₄	180000	510000	56000	96000	26000	52000	710000	4400000

Source: P:\01_SITES\Fording_River\1CT017.007_MLARD\400_Water Quality Predictions\412_Predict Loadings\Historic_Waste_Flushing_Terms\Historic_Waste_Flushing_Terms_1CT017.007_rev05_Id_AML.xlsx

3.4 Coal Reject Concentrations

Table 8 provides maximum observed concentrations in seepage from Greenhills Area A CCR Dump. The statistics were calculated based on three samples collected in 2009. These concentrations were used as a fixed concentration source term for coal reject.

Table 8. Coal Reject Fixed Concentrations

Parameter	Units	Value
Alkalinity	mg CaCO ₃ /L	490
Ca	mg/L	340
Cd	mg/L	<0.0003
F	mg/L	0.19
K	mg/L	5.2
Mg	mg/L	190
Na	mg/L	9.8
NH ₃	mg N/L	0.14
NO ₂	mg N/L	<0.02
NO ₃	mg N/L	<0.1
pH	s.u.	7.4
Se	mg/L	0.0087
SO ₄	mg/L	1300

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3.5 Co-Disposed CCR and Waste Rock

Table 9 provides release rates for CCR based on humidity cell tests performed on CCR from LCO. CCR release rates are an order of magnitude higher than waste rock, with the finer particle size distribution of CCR likely an important difference. These rates were decreased by a factor of 0.3 to reflect lower site temperatures.

Due to the very small dataset, care must be taken in applying these rates to CCR deposits. If there is a need to evaluate the effect of codisposal on loadings, additional testing should be performed.

Table 9. Codisposed CCR Release Rates

Material	SO ₄	Se
	g/m ³ /year	mg/m ³ /year
CCR	61	33

Source: P:\02_MULTI_SITES\Elk_Valley_Coal_Corp\1CE003.001_Selenium_Geochemistry\Phase_3_Implementation\Laboratory Program\5.Results\HCTs\Outcomes\Elk Valley KT Outcomes 1CE003.001 mcc REV00.xlsx]

4 Discussion

4.1 Relationship between Waste Rock Quantity and Selenium Release

The underlying assumption in the method is that chemical loadings of soluble ions generated by weathering of rock are primarily a function of waste rock volume. This assumes that leaching of these ions is not generally controlled by solubility because that would result in loadings being a function of contact water volume and consequently the approximate facility area rather than volume. The basis for this decision was a result of the following observations:

- The observation that selenium concentrations in the Elk River have increased roughly in parallel with the increase in cumulative spoil volume (Figure 1)
- The presence of sulphate concentrations in drainage from relatively undiluted seeps, and baseflow drainage at concentrations lower than required to precipitate gypsum
- The presence of a predictable linkage between rates of leaching observed at small scale and those at full scale (Kennedy et al. 2012)
- From a geochemical perspective, the narrow range in release rates calculated for spoils with volumes ranging over two orders of magnitude

4.2 Effect of Spoil Aging on Release Rates

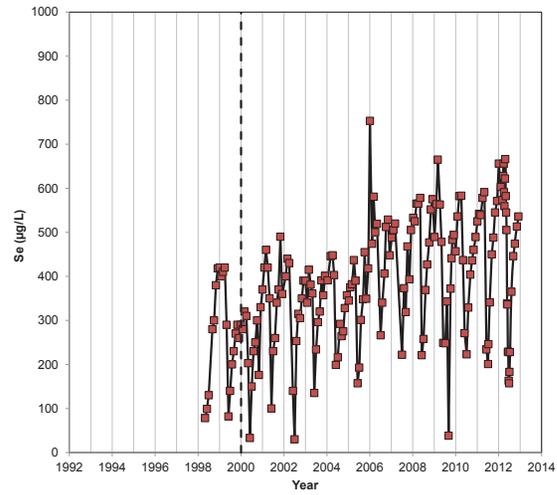
There are few sites in the Elk Valley that allow the impact of aging to be effectively evaluated because most spoils continue to be active. In the current assessment, LCO WLC and EVO SM1 monitoring sites represent spoils that have had limited or no recent spoiling activity.

The drainage from WLC has not yet shown a decrease in selenium or sulphate concentrations since major spoiling activities ended in 2000 (Figure 102). Nitrate concentrations are decreasing, which is consistent with explosives residuals being leached. Likewise, SM1 has not shown a decrease in concentrations even though monitoring of selenium and sulphate began 20 years after the dump was constructed (Figure 123). It is expected that any decrease would follow an exponential trend, with a relatively rapid initial decrease followed by a flattening decay trend.

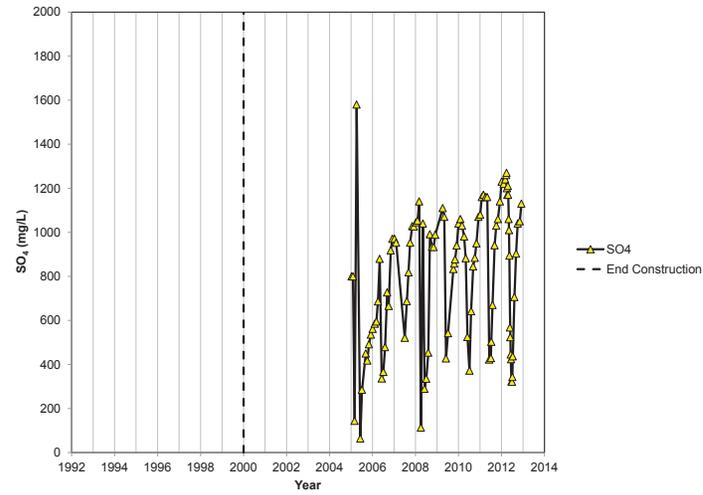
Using the average age parameter, only the nitrate release rate decreased with age (Figure 6), which was also observed at WLC. SM1 showed the lowest average selenium release rate in the dataset. SM1 is also the oldest site, and the age of the site is potentially a factor in the low selenium release rate. For example, long-term breakdown of the spoil and reclamation may be allowing sub-oxic conditions to develop in the spoil.

4.3 Calculation of Nitrate Loadings

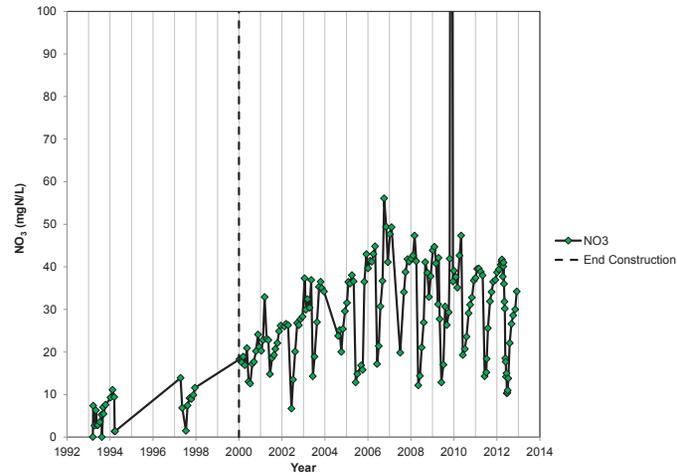
The equations shown in Table 3 were initially applied to nitrate loading in water quality models developed for the Elk Valley, with \bar{t} calculated based on waste schedules. Due to the exponential form of the equations, when the average age is near one year, nitrate loadings calculated using this equation were found to result in very high and unrealistic nitrate concentrations in waters. Use of these equations was therefore replaced with the published Environment Canada method described by Ferguson and Leask (1988) for operational conditions. Their method was developed based on interpretation of monitoring data in the Elk Valley and uses the powder factor and types of explosives as inputs. The equations have been applied to decay of nitrate concentrations after waste rock placement ceases.



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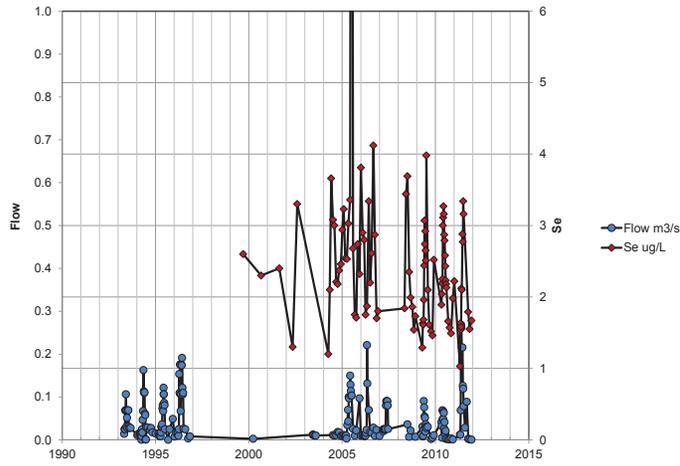


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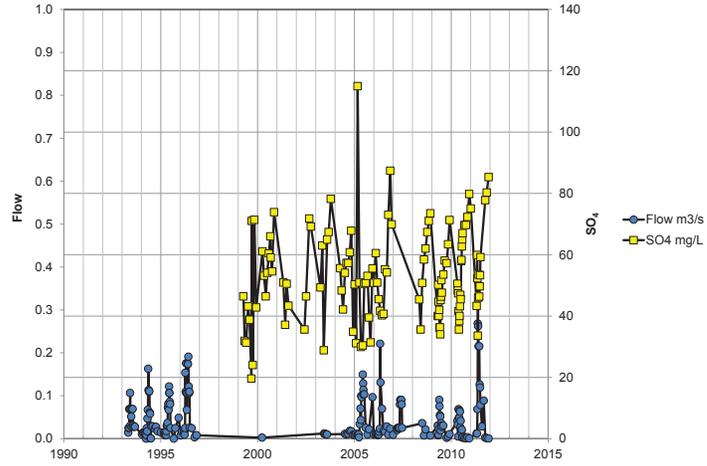


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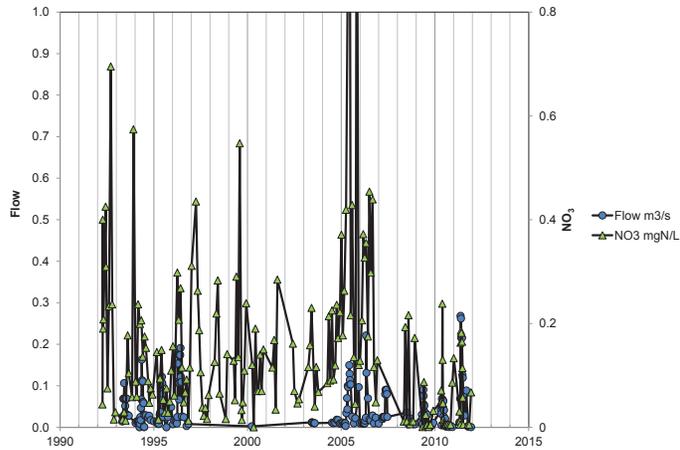
Figure 12. Drainage Chemistry for WLC



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Figure 13. Drainage Chemistry for SM1.

4.4 Application of Release Rates to the Elk Valley Water Quality Model

As described in Section 2.1, the source-term method was developed to provide a means for predicting average source term water chemistry for future spoils. Application of the average rates in Table 3 and Table 4 and average seasonal distribution of loadings to existing spoils requires the use of calibration factors to match simulated concentrations to historical data. Because the source-term method is based on the range of conditions throughout the valley, calibration factors are inevitably required because rates do vary, albeit over a range that is fairly narrow from a geochemical perspective.

A number of potential explanations for the variability in rates can be hypothesised:

- Measurement accuracy:
 - Completeness of monitoring record
 - Quantification of spoil volumes in catchment areas
 - Delineation of catchment areas
 - Flow measurement methods
- Geochemical factors:
 - Variation in the geochemical characteristics of the host rocks, including variations in absolute concentrations of reactive minerals and ratios of reactive minerals
 - For selenium, effects of processes that can cause attenuation, such as the presence of saturated fills, partially saturated zones in ex-pit spoils, and configuration of spoils affecting oxygen availability, including lift heights, traffic surfaces, and reclamation
 - Depletion of sources of reactive minerals
 - Spoil structure and stability over time (whether spoil failures have occurred)
 - Depletion of explosives residuals
- Hydrologic Factors:
 - Fraction of surface water collected at monitoring locations and loss to groundwater
 - Effect of hydrological variations on leaching

Since the influence of most of these factors cannot currently be quantified, the performance of any future wastes from new mining areas should be evaluated using rates shown in Table 3 and Table 4. As a potential future consideration, the simulation of current spoils receiving ongoing wastes from existing mining areas could be assessed using the rates observed for those spoiling areas, where adequate data are available. The current method of using calibration factors achieves the same objective. The limitation of this approach is that different calibration factors are likely to be needed for sulphate and selenium because there is some variation in ratios as shown in Figure 8.

5 Conclusions

This report summarizes the method used to develop geochemical source terms for sulphate, selenium, nitrate and cadmium from the waste rock spoils, CCR dumps and open pit walls at Teck's operations in the Elk Valley. The interpretations used to develop the source terms resulted in the following conclusions:

- Release of selenium and sulphate from waste rock spoils can currently be predicted using the cumulative volume of spoil.
- Release of nitrate from waste rock spoils can be predicted using the volume of spoil and the average age of the spoil once waste placement stops. The method does not appear to be appropriate for relatively young operational spoils for which the Ferguson and Leask (1988) has been used.
- Selenium and sulphate release rates calculated on a spoil volume normalized basis vary over less than an order of magnitude.
- Monthly release rates of selenium and sulphate are variable. The highest release is observed during snowmelt conditions and lowest release occurs during the winter.
- Variation in selenium and sulphate release rates can probably be attributed to several factors related to measurement methods, geochemical factors, waste rock dump construction, reclamation methods, and hydrological factors. These factors are not readily quantifiable; and therefore, average rates and their confidence limits are the best representation of performance of historical wastes lacking monitoring data and future wastes.
- Release of cadmium from waste rock spoils should be predicted using a fixed concentration source term.
- Release of selenium, sulphate and cadmium from coal reject dumps should be predicted using fixed concentration source terms.

This report, "Geochemical Source Term Inputs and Methods for Elk Valley Water Quality Planning Model", was prepared by



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All data used as source material plus the text, tables, figures, and attachments of this document have been reviewed and prepared in accordance with generally accepted professional engineering and environmental practices.

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