REVIEW OF AMENDMENT APPLICATION SUBMITTED DECEMBER, 2013 FOR THE SNAP LAKE MINE

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LIST OF ACRONYMS

AEMP	Aquatic Effects Monitoring Program
ALL	Annual Loading Limit
AML	Average Monthly Limit
BLM	Biotic Ligand Model
BUAD	Bedrock Unit Above the Dyke
BUAD2	Bedrock Unit Above the Dyke – 2nd Layer
BUBD	Bedrock Unit Below the Dyke
CCC	Criterion Continuous Concentration
CCME	Canadian Council of Ministers of the Environment
CCREM	Canadian Council of Resource and Environment Ministers
COC	Contaminant of Concern
DBCI	De Beers Canada Inc.
DOC	Dissolved Organic Carbon
EAR	Environmental Assessment Report
EPA	Environmental Protection Agency
EQC	Effluent Quality Criteria
FAV	Final Acute Value
GEMSS	Generalized Environmental Modelling System for Surfacewaters
HC5	Hazardous Concentration for 5% of tested species
IC20	Inhibitory Concentration producing a 20% response
LC20	Lethal Concentration producing a 20% response
LOEC	Lowest Observed Effect Concentration
MAC	Maximum Acceptable Concentration
MATC	Maximum Acceptable Toxicant Concentration
MDL	Maximum Daily Limit
MOE	Ministry of Environment
MVEIRB	Mackenzie Valley Environmental Impact Review Board
MVLWB	The Mackenzie Valley Land and Water Board
SSD	Species Sensitivity Distribution
SSWQO	Site-specific water quality objectives
TDS	Total Dissolved Solids

WASP5	Water Quality Analysis Simulation Program – Version 5
WLA	Waste Load Allocation
WQO	Water Quality Objectives
WTP	Water Treatment Plant

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1.0 Background

The Mackenzie Valley Land and Water Board (MVLWB) has received a request from De Beers Canada Inc. (DBCI) to amend water licence MV2011L2-0004 for the Snap Lake Diamond Mine. This request proposes a number of new site-specific water quality objectives (SSWQOs) and related effluent quality criteria (EQCs), including a revised SSWQO for total dissolved solids (TDS) to replace the value that was established during the original environmental assessment approved by the Mackenzie Valley Environmental Impact Review Board (MVEIRB), and EQCs for chloride, fluoride and nitrate to replace values that were prescribed by the MVLWB in 2012/13. The MVLWB has asked EcoMetrix to prepare an independent review of relevant DBCI documents supporting the requested amendment, focusing on a number of specific questions, listed below.

- *Are the proposed water quality objectives (WQOs) appropriate for the aquatic receiving environment?*
- *Are any of the contaminants of concern likely to exceed water quality objectives (WQOs) in the aquatic receiving environment?*
- *For those contaminants that are expected to exceed WQOs, what are the potential effects to aquatic life in Snap Lake and the downstream receiving environment?*
- *Are there feasible mitigation measures that can be implemented at the Snap Lake mine site that will either ensure that contaminants do not exceed WQOs, or will minimize effects to the aquatic receiving environment?*
- *Is the proposed method of calculating EQC appropriate to meet the dual objective of minimizing waste discharge and protecting downstream water uses?*

1.1 Documents Reviewed

In addressing the specific questions posed, as outlined in the terms of reference, the following documents submitted by DBCI to the MVLWB were reviewed:

- *TDS Response Plan (DBCI, Dec. 2013).* Development of TDS benchmark in attachment 2, development of a fluoride benchmark in attachment 1, and rationale for a chloride SSWQO in section 3.1.2.
- *Nitrogen Response Plan (DBCI, Dec. 2013).* Development of a nitrate benchmark in attachment 1, rationale for an ammonia SSWQO in section 3.1.2.
- *Strontium Response Plan (DBCI, Dec. 2013)*. Rationale for a strontium SSWQO in section 4.

- *Evaluation of EQC Report (DBCI, Dec. 2013).* Rationale for SSWQOs for several metals with hardness-dependent WQOs in Appendix 1.

- *Groundwater Flow Model Update (Itasca Denver, Aug. 2013)*. Prediction of the quality of water coming from the underground mine.

- *Mine Site Water Quality Update (DBCI, Dec. 2013)*. Prediction of the quality of site discharge from now to end of mine life.

- *Snap Lake Hydrodynamic and Water Quality Model Report (DBCI, Dec. 2013)*. Prediction of long-term water quality in Snap Lake using information from the groundwater and site water reports.

- *2012 Plume Characterization Study (Golder Associates, Jan. 2013)*. Monitoring of receiving water quality in 2012 to characterize the plume following installation of a new outfall diffuser in 2011.

1.2 Organization of Report

This report presents our review of information provided relevant to the questions posed. The questions are addressed in order, in Sections 2 through 6 below. References cited are in Section 7.

2.0 ASSESSMENT OF WATER QUALITY OBJECTIVES

Are the proposed water quality objectives (WQOs) appropriate for the aquatic receiving environment? The Board needs to decide what the appropriate WQOs should be for Snap Lake. The MVLWB chose a set of site specific water quality objectives (SSWQOs) for the project during the renewal of the Snap Lake water licence in 2012/2013; however, DBCI has proposed several changes. The following report sections evaluate the suitability of proposed WQOs, and outline factors the Board may wish to consider when making a determination about the proposed WQOs and SSWQOs.

2.1 Total Dissolved Solids (TDS)

Total dissolved solids (TDS) includes a variety of major ions, of which calcium and chloride are the dominant types at present in Snap Lake, accounting for 20% and 45% of TDS in 2012. The ion mixture has been relatively constant in Snap Lake since 2008. Since TDS effects on aquatic life depend on ionic composition, and because most of the toxicity literature on TDS pertains to ion mixtures different from those at Snap Lake, site-specific toxicity testing was commissioned by Golder and performed by Nautilus (2012a) on behalf of DBCI to determine low chronic effect levels relevant to TDS in Snap Lake. The test species included two alga (*P. subcapitata* and *N. pelliculosa*), a rotifer (*B. calyciflorus*), two daphnid cladocera (*C. dubia* and *D. magna*), a midge (*C. dilutus*) and lake trout (*S. namaycush*). The chronic test endpoints included survival, growth and reproduction.

TDS concentrations from 132 mg/L up to 1500 mg/L (nominal) were tested. Only the two daphnid tests gave a clear dose response (indicative of adverse effects). For *C. dubia*, the IC10 and IC20 for reproduction were 560 and 778 mg/L. For *D. magna*, IC10 and IC20 for reproduction were 312 and 684 mg/L. The lowest IC20 (684 mg/L) was selected as the proposed SSWQO (DBCI, 2013a).

The use of an IC20 level for a sensitive daphnid species as the SSWQO seems reasonable. The 20% response level is a level at which effects often become statistically discernable or measurable in both laboratory and field studies (Suter 1995). It represents a threshold level for effect. A lower response level such as the calculated IC10 might be considered. It would be more precautionary, representing a level with no measurable effect. DBCI (2013a) notes that daphnids are a minor part of the zooplankton community in Snap Lake (baseline and present day) and in Northeast Lake (a reference). Copepods and rotifers are the dominant zooplankton taxa. The rotifer tested was tolerant to TDS based on lack of dose-response; however, based on AEMP data (Golder, 2012) it is not a Snap Lake species. The resident species were not tested since toxicity test protocols are lacking.

The TDS tolerance of the dominant zooplankton in Snap Lake was not been well studied. Galat and Robinson (1983) studied two copepod species, present in Pyramid Lake, NV, at a TDS level of 5,350 mg/L (mainly NaCl), and found reduced abundance of the cyclopoid copepod *C. vernalis* in mesocosms when TDS was increased to 8,500 mg/L. Derry et al.

(2003) studied the densities of zooplankton species in lakes over a wide range of salinity. They found that most cyclopoid copepods, like cladocera, had their highest densities in sub-saline lakes (TDS <3,000 mg/L) while some other copepod species seemed to prefer higher salinity. Similarly, most rotifers preferred sub-saline conditions, with a few exceptions.

The two alga tested showed growth stimulation relative to the control, at all test concentrations, with somewhat less stimulation at the highest test concentrations. The laboratory control water had around 200 mg/L TDS (ion composition not tailored to Snap Lake). The stimulatory effect would be consistent with a TDS contribution to increased algal (phytoplankton) biomass observed in Snap Lake in AEMP monitoring (Golder, 2012). TDS was initially very low in Snap Lake but now approaches 300 mg/L in the diffuser area and 250 mg/L in the main basin. The EAR predicted a slight increase in phytoplankton biomass in Snap Lake, as noted by Golder (2012).

The EAR predicted that TDS would increase, up to a maximum whole-lake average of 350 mg/L, which is the basis of the existing licence limit. However, the whole-lake average has been increasing more rapidly than predicted, as further discussed in Section 3; hence the desire for a higher effects-based level. The proposed SSWQO of 684 mg/L is in the range of 500 – 1000 mg/L used for permitting by the State of Alaska (EAR, 2012) and seems appropriate as an adequately protective level for Snap Lake based on the site-specific toxicity testing.

2.2 Chloride

Chloride is a major component of TDS in Snap Lake, accounting for 45% of TDS in 2012. The CCME (2011) water quality guideline (WQG) for protection of aquatic life is 120 mg/L. Concentrations in the diffuser area reached this level in 2012 and are only slightly lower in the main basin. Similar to TDS, the concentrations are trending up more rapidly than expected, as further discussed in Section 3.

DBCI (2013a) has proposed a hardness-dependent SSWQO derived by Elphick et al. (2011a) for the Ekati site, and approved by the Wek'eezhii Land and Water Board (2013), as follows:

SSWQO (mg/L) = 116.6 ln(H) -204.1 (H = hardness mg/L as $CaCO_3$ up to 160)

Hardness is capped at 160 mg/L because the hardness effect was not investigated at higher levels. Thus, the maximum SSWQO would be 388 mg/L. Hardness was around 140 mg/L in the main basin of Snap Lake in 2013. The SSWQO was developed by adjustment of an equation describing the hardness-dependence of IC25 for the daphnid *C. dubia*. The adjustment factor was derived as the ratio of the 5th percentile from a species sensitivity distribution (SSD) to the IC25 for *C. dubia* at the same hardness. The adjustment served to reduce the SSWQO slightly from the value given by the equation for *C. dubia* IC25 vs water hardness. Elphick's SSD was based on data for 15 species (2 fish, 9 invertebrates and 4 alga/ plants).

The CCME (2011) recognized that chloride toxicity is hardness-dependent for many species, but decided insufficient data were available to develop the relationship for chronic toxicity. However, the CCME indicates that jurisdictions may choose to derive site-specific hardness-adjusted water quality objectives where appropriate. Based on Elphick's work, it is reasonable to infer a chronic toxicity - hardness relationship for the most sensitive species tested (daphnid invertebrates). When studied for other species, a hardness effect has usually been evident, although many of the studies are short-term.

The CCME (2011) also used an SSD approach, and the most sensitive species in that dataset were several species of mussels (not found in Snap Lake). The next most sensitive species in the CCME dataset is a fingernail clam (*Musculium securis*), with a reported LOEC for natality of 121 mg/L as chloride, using NaCl as a toxicant (Mackie, 1978). The study was not used or mentioned by Elphick. It did not follow a standard test protocol and did not use measured toxicant concentrations, but had controls and produced a good dose response. A hardness effect is suggested by the much higher LOEC of 756 mg/L using $CaCl_2$ as a toxicant. Fingernail clams have been increasing in effluent-exposed areas of Snap Lake (Golder, 2012). Thus, they do not seem to be adversely affected at present, and are unlikely to be so in future if the SSWQO of 388 mg/L is achieved.

The proposed SSWQO for TDS would actually limit chloride to a slightly lower level if the chloride ion contribution to TDS remains in the 45 to 47% range where it has been since 2010. Overall, the proposed SSWQO for chloride seems appropriate as an adequately protective level for Snap Lake under discharge conditions.

2.3 Fluoride

Fluoride is a very minor component of TDS in Snap Lake, with concentrations up to 0.2 mg/L in the diffuser area. The CCME (2002) water quality guideline (WQG) for protection of aquatic life is 0.12 mg/L. Fluoride began to exceed this level in the main basin in 2009. Concentrations are trending up, and the trend is expected to continue, but not to exceed 0.5 mg/L based on recent modelling (DBCI, 2013a).

DBCI (2013b) has proposed a SSWQO of 2.46 mg/L, derived as the 5^{th} percentile of a chronic species sensitivity distribution (SSD), following CCME (2007) protocol. The SSD was based on chronic test data for 15 species (3 fish, 8 invertebrates and 4 alga/plants). Endpoints were typically IC10 or IC25, but were constrained by what the study reported, including one LC50 value for the zebra mussel.

Water hardness has often been reported as a toxicity modifying factor, but this seems to be inconsistent among species. The lowest endpoint from each study was used, which meant using the soft water value (lower fluoride) in several cases where multiple hardness levels were tested. In one case, use of the higher hardness result would not have substantially changed the SSD; in the other case the higher hardness result was suspect because precipitation was observed.

The CCME (2002) guideline of 0.12 mg/L is an interim value based on a 6-day LC50 of 11.5 mg/L for caddisfly larvae (Camargo et al., 1992) with a 100-fold safety factor. This safety factor is quite conservative, resulting in a WQG well below the range of known effect levels. The SSD approach is preferable given the sufficient chronic toxicity database now available.

Only one organism in the database used by DBCI (2013b) had an endpoint below the proposed SSWQO (the fingernail clam with a MATC of 2.25 mg/L, Sparks and Sandusky, 1983). One lower test result not used was a 10-day LC10 for juvenile rainbow trout (1.8 mg/L at $18^{o}C$, 2.2 mg/L at $13^{o}C$, Angelovic et al., 1961), rejected because of its sub-chronic duration. The $18^{o}C$ value may not be site-relevant due to its high temperature.

Overall, the proposed SSWQO for fluoride seems to be adequately protective for aquatic life in Snap Lake. A lower target might be accommodated in the interest of non-degradation, based on lake water quality predictions.

The maximum acceptable concentration (MAC) for drinking water is 1.5 mg/L (Health Canada, 2012), based on moderate dental fluorosis at higher levels (an aesthetic effect). This should not be exceeded at the drinking water intake.

2.4 Nitrate

Nitrogen releases to Snap Lake (as nitrate and ammonia) originate from explosives residue, either from underground to the water treatment plant, or from the North Pile tailings to the water management pond. Ammonia undergoes biodegradation to nitrate through the action of nitrifying bacteria. Nitrate concentrations in 2012 reached 3 mg/L as NO_3-N in the diffuser area and 2.5 mg/L in the main basin. The CCME (2012) water quality guideline for NO_3-N is 2.93 mg/L. Concentrations are trending up, and the trend is expected to continue, reaching up to 9 mg/L in the diffuser area and 8 mg/L in the main basin based on recent modelling (DBCI, 2013c, Figure 3-3).

DBCI (2013c) has proposed a hardness-dependent SSWQO for NO_3-N, derived by Rescan (2012) for the Ekati site, and approved by the Wek'eezhii Land and Water Board (2013), as follows:

SSWQO (mg/L) = $e^{(0.9518 \ln(H) - 2.032)}$ (H = hardness mg/L as $CaCO_3$ up to 160)

Hardness is capped at 160 mg/L because the hardness effect was not established at higher levels. Thus, the maximum SSWQO would be 16.4 mg/L. Hardness was around 140 mg/L in the main basin of Snap Lake in 2013. The SSWQO was derived by developing a species sensitivity distribution (SSD) of threshold effect levels across species, adjusted to a common soft water hardness (40 mg/L), finding the 5th percentile of the distribution (HC5), and describing how the HC5 is expected to increase with water hardness. The soft water HC5 was 4.39 mg/L and the equation for adjusting it was:

SSWQO (mg/L) = $4.39\ e^{0.9518\ [\ln(H) - \ln(40)]}$

This equation simplifies to the first equation above. The slope of the hardness relationship was derived as a pooled slope using chronic IC25 data for four species (fathead minnow, amphipod *H. azteca*, midge *C. dilutus* and daphnid *C. dubia*) (Nautilus, 2011a). A hardness effect was suggested also for the rainbow trout MATC (Nautilus, 2011b) but these data were not utilized. A later study with lake trout (Nautilus, 2012b) also showed a hardness effect.

Rescan (2012) followed CCME (2007) protocol for the SSD approach. The test species included 4 fish, 4 invertebrates and 1 plant, from two Nautilus studies and five published studies. Toxicity endpoints included EC20, IC25 and MATC values. Some taxa from the literature (amphibians, decapods) were excluded because they did not occur locally.

To evaluate the suitability of the Rescan SSWQO for Snap Lake, site-specific toxicity testing was commissioned by Golder and performed by Nautilus (2013a) on behalf of DBCI to determine low chronic effect levels relevant to nitrate in Snap Lake. The tests were performed using the second and third most sensitive species from the Rescan SSD, the daphnid invertebrate *C. dubia* and the fathead minnow, tested in synthetic Snap Lake waters at two hardness levels (140 and 350 mg/L as $CaCO_3$) representing current and future minimum hardness. Major ion composition was tailored to Snap Lake: 21% calcium, 50% chloride, 12% sodium and 9% sulphate. The chronic test endpoints included survival, growth and reproduction.

The combined low effect levels from Rescan (2012) and the IC25 values using Snap Lake waters are plotted in Figure 3-4 of DBCI (2013c), along with the proposed SSWQO as a function of hardness.

All the low effect levels fall above the proposed SSWQO, suggesting that the SSWQO is protective. The daphnid *C. dubia* was more sensitive than the fathead minnow in Snap Lake water. The IC20 for *C. dubia* reproduction was 26 mg/L NO_3-N at hardness 140 mg/L and 16.7 mg/L NO_3-N at hardness 350 mg/L. This latter value was just marginally above the SSWQO value of 16.4 mg/L at high hardness.

The lower response levels for *C. dubia* at hardness 350 mg/L as compared to 140 mg/L raise a question as to whether nitrate response levels may be even lower at the upper bound of future hardness around 950 mg/L. Nitrate toxicity was not tested at this level of hardness.

Toxicity test data are not available to address nitrate toxicity to rotifers and copepods, the dominant zooplankton taxa in Snap Lake. It is hoped that a SSWQO protective of daphnids would also protect these other invertebrate taxa, since daphnids are often among the more sensitive test species. However, the nitrate sensitivity of the dominant zooplankton taxa remains uncertain.

The toxicity testing does not consider nutrient enrichment effects. There have been both nitrogen and phosphorus inputs to Snap Lake as a result of mine operations (Golder, 2012). Nitrogen in lake water is increasing as noted above. Phosphorus measured with the plankton component of the AEMP has increased up to 2009, with subsequent decline. Phytoplankton biomass has followed a similar pattern (Golder, 2012, Figure 3.2-9). Nitrogen:phosphorus ratios were optimal for phytoplankton growth in 2006 and 2007, but indicate phosphorus limitation since 2008, and severe limitation in the main basin in 2010 and 2011. Thus, while nitrogen has likely contributed to algal growth stimulation, the further nitrogen increases at this stage may be contributing to a reverse effect. The EAR predicted a slight increase in phytoplankton biomass in Snap Lake, as noted by Golder (2012).

Overall, the proposed SSWQO for nitrate seems to be adequately protective against toxic effects in Snap Lake, but there is some uncertainty as to protective levels of nitrate under future hard water conditions, particularly for upper bound hardness scenarios. This could be resolved by additional site-specific toxicity testing. A lower target for nitrate in the lake might be accommodated in the interest of non-degradation, based on lake water quality predictions.

The maximum acceptable concentration (MAC) for drinking water is 10 mg/L NO_3-N (Health Canada, 2012), based on concerns about gastrointestinal conversion of nitrate to nitrite and associated methemoglobinemia in infants. The MAC should not be exceeded at the drinking water intake.

2.5 Ammonia

Total ammonia includes both ionized (NH_4^+) and unionized (NH_3) forms. The unionized form is most toxic to aquatic life. The proportion unionized increases with both pH and temperature. The CCME (2010) water quality guideline is 0.019 mg/L for NH_3 (0.016 as N), and is a corresponding function of both pH and temperature for total ammonia. Total ammonia concentrations in 2012 reached 0.32 mg/L as N in the diffuser area, and around 0.2 mg/L in the main basin. They have been relatively constant since 2009, but are predicted to increase up to 2.5 mg/L in the diffuser area and around 2 mg/L in the main basin by 2028 based on recent modelling (DBCI, 2013c, Figure 3-8). Unionized ammonia in 2012 was up to 0.002 mg/L as N, and is predicted to be up to approximately 0.005 mg/L by 2028 (DBCI, 2013c, Figure 3-9).

DBCI (2013c, Section 3.1.2) has proposed to continue to use the CCME chronic guidelines (WQG) for both unionized and total ammonia. The latter was evaluated at 85th percentile values of pH and temperature (7.14 and 13.7^{o}C) to define a WQG (5.21 mg/L as N) on which to base calculation of an effluent quality criterion (DBCI, 2013c, Figure 3-8). The value of 5.21 mg/L seems to be wrong; using the CCME equations, and the reported 85th percentiles of pH and temperature, the correct value is 4.6 mg/L as N.

Meeting the CCME chronic guideline for total ammonia, over chronic averaging periods, should ensure that average unionized ammonia does not exceed 0.019 mg/L, the lower 95% confidence limit on the 5th percentile of the CCME species sensitivity distribution (SSD) for unionized ammonia. The U.S. EPA (2013) has used an SSD approach, but has identified some lower effect levels, particularly for unionid mussels. In the absence of these mussels, with fish early life stages present, the EPA criterion for pH 7.14 and 13.7^{o}C is about 6.2 mg/L as N. The lower CCME guideline is preferable and should be adequately protective in the pH and temperature range of Snap Lake.

An acute (short-term) objective of 21 mg/L from the U.S. EPA (2013) is proposed as a concentration to be allowed for short periods (1 hour according to the U.S. EPA). DBCI (2013c, Table 3-2) proposes to use 21 mg/L for both average monthly and maximum daily EQCs. This is intended to avoid acutely toxic effluent. DBCI (2013c, Section 3.2.2.3) recommends that existing EQCs for ammonia should be retained, at 10 and 20 mg/L for average monthly and maximum daily limits. The latter values are apparently intended as the final EQCs. They should allow the CCME guideline to be achieved in the lake.

The proposal to use a chronic WQG for ammonia from CCME seems to be adequately protective as regards chronic toxicity effects in Snap Lake, however, the calculated value for 85th percentile pH and temperature (5.21 mg/L) is incorrect. The correct value is 4.6 mg/L.

2.6 Strontium

Strontium is found in Kimberlite and is released in treated effluent to Snap Lake, with concentrations up to 0.75 mg/L in the diffuser area, and 0.65 mg/L in the main basin, measured as total strontium in 2012. There is no CCME water quality guideline (WQG) for protection of aquatic life. Strontium is very soluble and is expected to be mainly in dissolved form. Concentrations are trending up, and the trend is expected to continue, but not to exceed 4 mg/L in 2028 based on recent modelling (DBCI, 2013d).

DBCI (2013d,e) has proposed a SSWQO of 14.1 mg/L, derived as the 5th percentile of a chronic species sensitivity distribution (SSD), following CCME (2007) protocol. The SSD was based on chronic test data for 12 species (4 fish, 7 invertebrates and 1 alga). Endpoints were typically IC10 to IC20, but were constrained by what the study reported, including several EC50 or LC50 values. The two highest values in the SSD appear to be acute rather than chronic values (1 day exposure for salmon fry, 2 day exposure for planaria). Chapman (2014) removed these values from the SSD and calculated a revised SSWQO of 10.7 mg/L for strontium.

Two effect levels reported in the literature have been problematic since they are well below the range of other toxicity test literature for strontium. Birge at al. (1980) reported a 28-day LC50 for rainbow trout embryos of 0.025 mg/L, and Borgmann et al. (2005) reported a 7-day LC50 for amphipods of >1 mg/L. The latter value is open-ended and based on data that were not corrected for control response. Toxicity tests were commissioned by Golder and performed by Nautilus (2012c, 2013b) on behalf of DBCI to determine whether these results were reproducible. Following Environment Canada test protocols, Nautilus found LC50 and LC10 values of >157.5 mg/L and 75.2 mg/L for rainbow trout embryos in soft water, an LC50 of 176.8 mg/L for the amphipod, and an IC10 of 31.2 mg/L for growth inhibition in the amphipod. The LC10 and the IC10 values were appropriately used in the SSD for strontium, in preference to the results from Birge et al. (1980) and Borgmann et al. (2005).

Strontium is chemically similar to calcium, and uptake by fish is inversely related to the level of calcium in the water (Chowdhury and Blust, 2012). Thus, a hardness effect on toxicity is likely, as indicated by the Nautilus (2013b) tests with rainbow trout embryos (no strontium effect with hardness at 100 mg/L). However, hardness effects on toxicity have not been fully explored.

Strontium has been accumulating in Snap Lake sediments over time (DBCI, 2013d, Figure 4-4). The lake-wide mean in sediments is now approximately twice the baseline level. There is no sediment quality guideline for strontium. Sediment quality changes may be more persistent after closure than water quality changes. In the absence of sediment toxicity data, the implications of strontium accumulation in sediments are uncertain.

Overall, the proposed SSWQO of 10.7 mg/L for strontium seems to be adequately protective as regards toxic effects in Snap Lake. A lower target might be accommodated in the interest of non-degradation, based on lake water quality predictions.

2.7 Sulphate

Sulphate is a minor component of TDS in Snap Lake, accounting for 9% of TDS in 2012. There is no CCME water quality guideline (WQG) for protection of aquatic life. Sulphate concentrations are trending up, and the trend is expected to continue. The maximum was 19 mg/L in the diffuser area in 2011, and 17 mg/L in the near-field and far-field areas (Golder, 2012). Based on recent modelling, the maximum predicted concentration is 120 mg/L in the diffuser area and 118 mg/L at the Snap Lake outlet.

DBCI (2013f) has proposed to adopt the B.C. MOE (2013) hardness-dependent water quality guideline as a SSWQO. This guideline, defined as a 30-day average concentration, varies as a step function of hardness:

128 mg/L at hardness 0 to 30 mg/L

218 mg/L at hardness 31 to 75 mg/L

309 mg/L at hardness 76 to 180 mg/L

429 mg/L at hardness 181 to 250 mg/L

The step function is based on the hardness relationship for sulphate toxicity to rainbow trout in a 21-day eyed embryo to alevin test (LC20 data) adjusted down by a 2-fold safety factor. This test was chosen as the most sensitive species and endpoint. The guideline was verified by comparison to chronic test data from other studies for 10 species (3 fish, 1 amphibian, 4 invertebrates, 2 alga/plants) tested at different hardness levels between 40 and 320 mg/L. All endpoints for 20-25% response fell at or above the guideline, indicating that the guideline was adequately protective in the studied range of hardness.

For hardness above 250 mg/L, based on concern about osmotic effects due to combined sulphate and hardness, it was recommended that further testing with site water would be needed to develop a site-specific sulphate guideline. It does not appear that such testing has been done for Snap Lake water. Thus, it is unclear whether the SSWQO of 429 mg/L for sulphate would be adequately protective at a future hardness of 350 to 950 mg/L. The Elphick et al. (2011b) data for hardness at 320 mg/L suggest that it may be protective at the low end of future hardness, since 25% response levels were well above 429 mg/L for most species. The lowest IC25 for hardness at 320 mg/L, for *C. dubia* reproduction (425 mg/L), was essentially at the proposed SSWQO level. Sulphate effect levels at the high end of future hardness are not known; they may decline at hardness above about 250 mg/L. There is some suggestion of this when we compare effect levels for *C. dubia* between hardness levels of 160 and 320 mg/L.

Stekoll et al. (2003) reported effects on fertilization in salmon at TDS (primarily $CaSO_4$) levels as low as 250 mg/L (around 176 mg/L SO_4). B.C. MOE (2013) recommended further research on effects of sulphate and hardness on fertilization and pre-eyed embryos of rainbow trout as these life stages are identified as being more sensitive than eyed embryos.

Overall, the proposed SSWQO for sulphate seems to be adequately protective against toxic effects in Snap Lake, but there is some uncertainty as to protective levels of sulphate under future hard water conditions, particularly for upper bound hardness scenarios. This uncertainty should be resolved by site-specific toxicity testing.

2.8 Copper

Copper in Snap Lake water is not correlated with conductivity as an effluent indicator, and does not show an increasing trend (Golder, 2012, Table 3.1-14). The maximum predicted concentration is 2.2 ug/L. The current CCME water quality guideline for copper in ug/L is a hardness-dependent equation (CCREM, 1987):

$$\text{WQO } (\mu g/L) = e^{(0.8545 \ln(H) - 1.465)} * 0.2 \qquad (H = \text{hardness mg/L as } CaCO_3 \text{ up to } 180)$$

The equation is referenced to an earlier U.S. EPA (1985) guideline, but is reduced by a safety factor of 0.2, based on perceived uncertainty in the chronic hardness relationship.

The slope of the hardness relationship is based on acute toxicity studies. The U.S. EPA water quality criterion has since been updated (U.S. EPA, 1996, 2007). The latest U.S. EPA guidance is to use the Biotic Ligand Model (BLM). The BLM estimates a final acute value (FAV) as the 5th percentile of the SSD for acute exposure, considering hardness and dissolved organic carbon (DOC) as toxicity modifiers, and then estimates a chronic criterion (CCC) as the FAV divided by an acute-chronic ratio of 3.22.

DBCI (2013f) has proposed to use an SSWQO in ug/L calculated from the hardness relationship in the EAR (DBCI, 2002):

$$SSWQO = (H/50)^{0.8545} * (7.9/ (180/50)^{0.8545})$$

This equation produces a SSWQO of 6.4 ug/L at H=140 mg/L, and 13.9 ug/L at H=350 mg/L.

We have not examined the derivation of this relationship; however, it closely approximates the U.S. EPA (1985) chronic criterion times 0.405, or roughly twice the CCME water quality guideline. It is roughly half the value of the U.S. EPA (1996) chronic criterion, which approximates the BLM-based criterion for standard soft to hard laboratory test waters with a DOC of 2 mg/L (U.S. EPA, 2007). The U.S. EPA (1996) chronic criterion is 12 ug/L at H = 140 mg/L and 27 ug/L at H = 350 mg/L.

A more realistic SSWQO might be obtained using the BLM with Snap Lake water chemistry; however, the proposed SSWQO is likely to be adequately protective based on comparison to BLM-based criteria for standard waters. Given a maximum observed copper concentration of 3.8 ug/L (possible sample contamination), and lake average values from 0.53 to 0.87 ug/L (Golder, 2012, Table 3.1-13), and no increasing trend in copper, the proposed SSWQO is unlikely to be reached.

2.9 Nickel

Nickel in Snap Lake water is correlated with conductivity as an effluent indicator, and does show an increasing trend (Golder, 2012, Table 3.1-14). The current CCME water quality guideline for nickel in ug/L is a hardness-dependent equation (CCREM, 1987):

$$WQO\ (\mu g/L) = e^{(0.76\ \ln(H) + 1.06)} \qquad (H = \text{hardness mg/L as } CaCO_3 \text{ up to } 180)$$

The equation comes from the U.S. EPA (1980). The slope of the hardness relationship is based on acute toxicity studies. An estimate of the 5th percentile of the SSD for acute toxicity was adjusted to a chronic objective using an acute-chronic ratio of 19.4. The WQG is 123 ug/L at H = 140 mg/L and 248 ug/L at H = 350 mg/L. The U.S. EPA water quality criterion has since been updated based on a new 5th percentile and a new acute-chronic ratio (U.S. EPA, 1996):

$$WQO\ (\mu g/L) = e^{(0.846\ \ln(H) + 0.0584)}$$

This equation produces lower criteria of 69 ug/L at H = 140 mg/L and 151 ug/L at H = 350 mg/L.

DBCI (2013f) has proposed to use the WQG from CCME as a benchmark for Snap Lake. Since nickel concentrations in the lake are almost always below 2 ug/L (Golder, 2012, Figure 3.1B-27) they are unlikely to reach either CCME or current EPA guideline levels in Snap Lake.

3.0 ASSESSMENT OF MODELS

Are any of the contaminants of concern likely to exceed water quality objectives (WQOs) in the aquatic receiving environment? The Board needs to understand whether any contaminants of concern are likely to exceed the accepted WQOs for Snap Lake. In order to support this determination, this Section reviews the models used by DBCI to make long-term predictions of water quality in Snap Lake, describes how accurate or conservative they may be, and outlines key assumptions and uncertainties in the models.

DBCI has revised the water quality modeling for Snap Lake, the mine site, and the groundwater in the vicinity of the mine due to "unanticipated increases in underground flows and treated effluent discharges to Snap Lake" (DBCI, 2013h). As a result of these increases, the whole lake average concentrations of TDS and chloride appear to be increasing more quickly than originally predicted. The revised water quality modeling led DBCI to predict that without additional mitigations, TDS and chloride concentrations in Snap Lake will exceed their proposed SSWQOs in the years 2016 and 2017. Other parameters were not predicted to exceed in the lake.

Each model used by DBCI in this revised modeling exercise has been evaluated by EcoMetrix, and our conclusions concerning these models are presented in the Sections below.

3.1 Groundwater Flow Model

As part of the water licence amendment, the groundwater flow model used to estimate water inflow to the underground mine was updated. The updated groundwater model (termed the "August Model") completed by Itasca (August, 2013) included the incorporation of new data supplied by DBCI including geological data, flow rate and chemistry data. Specifically, the August Model included (Itasca, 2013a):

- Updated conceptual site model including new geological data on structure zones (faults) and hydraulic conductivity values decreasing with depth in bedrock above and below the dyke;
- revisions to the mine plan;
- inclusion of measured mine inflow rates and TDS concentrations; and,
- Additional sensitivity analyses to address uncertainties.

Calibration of the model was completed using measured TDS concentrations and flow data. Volumetric mixing approaches were then used to predict TDS concentrations in future mine water discharges. Overall, the groundwater model appears to be appropriate. However, a number of limitations were identified in the updated model including limited water level data and TDS measurement locations. In addition, the extent of structure zones and associated hydraulic conductivities are not known.

Updated Geological Site Model

The recent updates to the geological site model included additional data from DBCI on the presence of faults within the model boundary. However, full delineation of the Snap and Crackle faults has not been completed; these faults are assumed to extend to the model boundaries, which is a conservative assumption. The hydraulic conductivity values are the main variables in the model that control groundwater flow estimates, and therefore additional delineation of these faults should be completed to reduce uncertainty in the inflow predictions.

The groundwater model was also updated to include a decrease in hydraulic conductivity values with depth in bedrock above the dyke (BUAD) and below the dyke (BUBD). It is noted, however, that the hydraulic conductivity values used for the second layer of the bedrock above the dyke (BUAD2) are the highest values for the hydraulic unit, inconsistent with the stated relationship to depth. The assigned hydraulic conductivity values for each layer were determined through model calibration to match measured inflow rates to the mine. Although over 140 hydraulic conductivity measurements were made through hydraulic testing, there is no indication that the estimated hydraulic conductivity values for the BUAD and BUBD used in the model are consistent with any measured values from hydraulic testing of surface boreholes or underground drill holes within these represented layers. Correlation with measured hydraulic conductivity testing would decrease the uncertainty of the model.

TDS Concentration Estimates

Estimates of TDS concentrations in the mine water are determined through volumetric mixing of inflow water from the hanging wall and foot wall. Concentrations in the water from the foot wall were assumed to stay constant while TDS concentrations in water from the hanging wall were allowed to increase due to the contributions from Snap Lake and associated increased concentrations in the Lake.

Initial concentrations of TDS used in the model for inflow from both the foot wall and hanging wall were reportedly based on water quality measurements from 2012 and 2013 (Itasca, 2013a). Both the arithmetic and geometric mean of measured TDS concentrations in the water from the hanging wall and foot wall are higher in 2013 in comparison to 2012 (Figure 4; Itasca, 20113a). In Itasca (2013b; “October update”), it is reported that data from 2008 to 2013 were used to determine initial TDS concentrations in inflow water for the model. However, no assessment of concentration trends is provided. We understand that the October update provides the predicted TDS concentrations in the discharge, and that these updated predictions are utilized in the Mine Site model. Since the footwall concentrations are considered constant in the model, the selected TDS concentrations for the footwall also represent the future concentrations. An assessment of possible temporal trends in the TDS concentrations, particularly for the foot wall, should be provided to determine if the increased TDS concentration in the footwall inflow between 2012 and 2013 is reflective of a long-term trend. A concern is that the increasing trend of TDS concentrations may continue, and that TDS loadings may be underestimated in the model, particularly in the future.

For the August Model, both the arithmetic and geometric mean values from the 2012/2013 data were used during sensitivity analysis, and the predicted concentration of TDS in the mine water was highly dependent on the initial concentration of TDS in water from the footwall. Calibration of the model also showed good correlation of measured values with predicted values prior to 2009 when the arithmetic mean is used in the model. However, after 2009, a better correlation is seen with measured values when the model incorporated the geometric mean. No explanation is provided for this difference. The October update used arithmetic mean TDS values as inputs, which are higher than the geometric means (Figure 11; Itasca, 2013a). This may tend to overestimate TDS loadings in future, if the model based on geometric mean is really tracking the recent measured data better. The fact that measured TDS is not consistently in agreement with either model suggests uncertainty about model performance into the future.

Overall Model Assessment

Overall, the groundwater model appears to accurately represent current and historical inflows to the mine and to approximate current and historical TDS concentrations. However, uncertainties associated with the model including hydraulic parameters of the future mining areas and measured TDS concentrations are limitations that should be addressed going forward. Itasca (2013a) provides recommendations for further assessment, including further delineation of the Snap and Crackle Faults, additional borehole completion and hydraulic testing and increased TDS monitoring. Implementation of these recommendations would reduce uncertainties in the model. Further model updates should be completed as additional data are obtained.

3.2 Mine Site Model

The overall objective of the Mine Site Model was to predict the water quality of the discharge to Snap Lake from the Snap Lake Mine. The basis of the model is a mass balance of multiple sources which are interconnected using the GoldSim software (version 10.50). The sources considered in the model for each COC were the mine itself, the north pile, the mine site, the water treatment plant, and the water management pond. The output from the mine source was largely driven by the results of the Groundwater Model described in the previous Section. Snap Lake itself was modeled separately, but the Snap Lake Model connects to the Mine Site Model via inflow from Snap Lake to mine water, and mine water contribution to the effluent discharged to Snap Lake. The two models were run iteratively, ensuring that concentrations of COCs in Snap Lake were also represented in the Mine Site

Model. All of the sources were linked in GoldSim to allow mass and flow balance calculations across the system.

Model inputs were based on several data sources, including on-site monitoring data; records of mining, material processing, explosives use; laboratory kinetic test data; geochemical controls; measured flow data; inputs from hydrology, hydrogeology, and lake mixing models; the Mine Plan; and the waste materials and management plan. Four scenarios were run in the Mine Site Model, with differing lake and connate water flows and concentrations of TDS in connate water. The scenarios encompass a range of possible rates of COC discharge from the mine, which vary in time for each scenario over the life of the mine.

Calibration of the Mine Site Model was accomplished using base case groundwater flows from the Groundwater Model and geometric mean TDS concentrations in connate water to match historical monitoring data measured in three locations between 2011 and 2013. The calibration exercise demonstrated that additional mass loadings, having not been previously considered in prior versions of the model, were now needed to achieve good model agreement with the actual measured data described above..

DBCI did not directly tabulate the assumptions made in the development of the Mine Site Model in their report, but several modeling assumptions can be inferred from the provided text. Among them are that the four selected modeling scenarios bracket a reasonable range of flows and concentrations when compared to reality; that all potential sources and losses of COCs in the system have been accounted for or are negligible; that the source terms for the various COC sources contributing to the model are reasonable; and that the additional mass loadings required for calibration make physical sense. In light of available information, these assumptions appear to be reasonable for constructing a water quality model.

As far as model uncertainties are concerned, DBCI acknowledged several limitations in their Mine Site Model report. These limitations state that the modeling will be invalid if changes to the mine and/or site conditions such as deviations from the Mine Plan or installation or removal of operations occur; that groundwater inflows are uncertain; and that the system is complex and the model may not completely describe all of the intricacies of the system. We acknowledge these limitations and associated uncertainties in the model predictions. As noted in Section 3.1, uncertainty in groundwater flows may be addressed by further assessment, but the remaining uncertainties are not expected to be reduced in the future. As such, these uncertainties are considered to be reasonable. We note from the calibrations for the mine water and the north pile that strontium, sulphate, and TDS appear to be underpredicted in north pile discharge, and thallium appears to be underpredicted in mine water. We also note that several COCs in the final discharge seem to be slightly overpredicted by the Mine Site Model, including fluoride, iron, ammonia, and TDS, according to the results of the Mine Site Model calibration. Those parameters that appear to be overpredicted are considered to have been modeled conservatively, but those that are underpredicted are not. As the final discharge is ultimately the stream of concern, the reasonably good calibration in the discharge leading to realistic to slightly conservative results indicates that the Mine Site Model appears to fulfill its objective.

3.3 Snap Lake Hydrodynamic Model

The Snap Lake Hydrodynamic Model is based on several pieces of specialized modeling software (DBCI, 2013i). The 3-dimensional base model of Snap Lake was developed in the Generalized Environmental Modelling System for Surfacewaters (GEMSS), but the Modified WASP5 module from the US EPA was used for simulating nutrients and oxygen-related constituents such as dissolved oxygen and biological oxygen demand (BOD) in Snap Lake. The user defined constituent module in the Snap Lake Hydrodynamic Model was derived from the CE-QUAL-W2 model, which was used for simulating constituents that behave conservatively or settle in the water column in Snap Lake.

Three types of inputs to the Snap Lake Hydrodynamic Model were entered by DBCI: meteorological, consisting of temperatures, air pressure, and wind data; hydrological, consisting of flows of effluent, tributaries, and non-point sources; and chemical, consisting of site monitoring data. As part of this latter group, compound-specific parameters associated with chemical and biological reactions for nutrients were also inputs into the model.

DBCI calibrated the model to measured data from 2004 to 2012. A hydrodynamic calibration was undertaken first to ensure that a water balance was achieved and to attempt to match measured water surface elevations. DBCI used model default hydrodynamic parameter values at this stage, with the exception of the addition of sediment and ice/water heat exchanges, which were added to the model to correct the thermal profile. In addition, in order to correct the vertical momentum dispersion in the model, the Nikuradse mixing length was applied instead of van Karman mixing length. Both changes improved the hydrodynamic calibration in the model.

After this hydrodynamic calibration was complete, a chemical calibration was performed. Several changes were made to the default parameters in the model to correct the calibration for nitrogen species, phosphorus species, and phytoplankton, including the following revisions:

- The nitrification rate was changed from 0.02 to 0.01 per day;
- The denitrification rate was changed from 0.09 to 0.16 per day;
- The Michaelis constant for denitrification was changed from 0.1 to 0.2 g-O_2/m^3;
- The phosphorus to carbon ratio was changed from 0.025 to 0.015 g-P/g-C;

- The dissolved organic phosphorus mineralization rate was adjusted from 0.22 to 0.33 per day;
- The half-saturation constant for phosphorus mineralization was changed from 5 to 1.5 g-C/m^3;
- The ratio of carbon to chlorophyll a was changed from 70 to 20;
- The phytoplankton death rate was changed from 0.015 to 0.005 per day; and
- The phytoplankton settling velocity was changed from 0.05 to 0.02 m/d.

After nutrient calibration was achieved, the remaining chemical parameters were calibrated. At this stage, additional mass loadings were added to the model so that barium was not underpredicted, and settling from the water column was added to the model to correct overpredictions of aluminum and uranium.

In the calibration step, DBCI also noted several potentially irregular model results that were not corrected. For instance, DBCI noted that fluoride was conservatively overpredicted in 2011 and 2012, whereas magnesium was "slightly" underpredicted, but no changes were made to the model calibration to account for these irregularities. For metals and metalloids, arsenic, copper, mercury, and zinc were noted to be overpredicted. DBCI concluded that the overpredictions were conservative, and did not comment further on the "slight" underprediction of magnesium. We concur with DBCI's implicit comment that this underprediction is immaterial.

As in the case of the Mine Site Model, the Snap Lake Hydrodynamic Model assumptions were not directly tabulated by DBCI. Nevertheless, two assumptions were inferred from the text. The first is that since the time series derived by DBCI for all of the source terms appear to rely on interpolation between monitoring samples, this interpolation was assumed to represent the actual behaviour of the source. In other words, there were no "spikes" or "troughs" in concentrations between successive samples that were not captured by the monitoring program, especially as far as effluent is concerned. The second is that transport processes not considered in the model were unimportant and did not materially affect the results of the modeling, a point which was discussed in detail by DBCI with respect to fluoride solubility controls and mercury transformations in Snap Lake. Both of these assumptions are reasonable for the current modeling exercise. DBCI documented two major uncertainties in the Snap Lake Hydrodynamic Model. The first is a collection of data-related uncertainties for each COC: DBCI described their long-term predictions of COC concentrations in Snap Lake as either moderately or highly uncertain due to uncertainties in the upstream models. Predictions for zinc, copper, arsenic, mercury, antimony, and orthophosphate during the calibration period were also considered highly uncertain but DBCI noted that these COCs were projected to remain well below WQOs, so this uncertainty is acceptable in the context of the current evaluation. DBCI also described a limitation of the model, which, as in the case of the Mine Site Model, was invalidation of the modeling if changes to the mine and/or site conditions such as deviations from the Mine Plan or installation or removal of operations occur. Again, we acknowledge that this limitation results from construction of a model based entirely on observed data and phenomena, and it is thus considered acceptable for this modeling exercise.

The Snap Lake Hydrodynamic Model appears to be accurate in its predictions of COCs in Snap Lake over the calibration period. No large underpredictions or overpredictions were observed in its results for those parameters which may meet or exceed WQOs. The degree of conservatism in the model, however, is not readily apparent; the predicted COC concentrations for future time periods are significantly higher than in previous modeling efforts, but these predictions may be realistic rather than conservative if current trends in effluent quality continue.

3.4 Overall Assessment of Models

In general, we are confident that the models are as accurate as possible given the data that DBCI has collected, and that the model predictions for future concentrations are either realistic or conservative. We concur with Itasca (2013a) that further assessments would reduce the amount of uncertainty in the groundwater characterization and modeling. Given that the groundwater, entering the Mine Site Model from the mine and then entering the Snap Lake Hydrodynamic Model through the water treatment plant, is an important driver of TDS concentrations, reductions in groundwater uncertainty are expected to reduce overall uncertainty in all linked models.

4.0 ASSESSMENT OF EFFECTS TO AQUATIC LIFE

For those contaminants that are expected to exceed WQOs, what are the potential effects to aquatic life in Snap Lake and the downstream receiving environment? DBCI has predicted that without additional mitigations TDS and chloride concentrations in Snap Lake will exceed the proposed SSWQOs as early as 2015 to 2016. Although DBCI has proposed some potential mitigations, their effectiveness is uncertain. The following sections discuss potential effects of the proposed exceedances, and whether they would constitute a significant adverse effect.

4.1 Total Dissolved Solids (TDS)

DBCI (2013g) has predicted that TDS will exceed the SSWQO of 684 mg/L in the diffuser area as early as 2015 for the worst-case scenario (upper bound inflow, arithmetic mean TDS) and by 2018 for the best-case scenario (lower bound inflow, geometric mean TDS). Predicted peak concentrations will be about 1700 mg/L in Snap Lake for the worst-case scenario and 820 mg/L for the best-case scenario. Peak concentrations in the main basin will be only slightly lower. The EAR predicted that TDS concentrations in would fluctuate around 350 mg/L in the diffuser area, and not exceed this level in the main basin.

Site-specific toxicity tests are the best indicator of potential effects from elevated TDS, because they reflect the ion mixture that is expected in Snap Lake under discharge conditions. The two daphnid tests (*C. dubia* and *D. magna*) showed a dose response to TDS in the tested concentration range. For *C. dubia*, the IC10, IC20 and IC50 for reproduction were 560, 778 and 1368 mg/L. Therefore, at the worst-case prediction of 1700 mg/L in Snap Lake, we would expect more than 50% inhibition of reproduction. In the highest tested concentration (1474 mg/L) reproduction was reduced to 42% of the control value. For *D. magna*, the IC10, IC20 and IC50 for reproduction were 310, 684 and >1474 mg/L. Therefore, at the worst-case prediction of 1700 mg/L, we would expect at least 20% inhibition of reproduction, and probably more. Extrapolation beyond the range of test concentrations is uncertain.

For the other invertebrates tested (rotifer *B. calyciflorus*, midge *C. dilutus*) and the fish tested (lake trout, arctic grayling) there was no dose-response up to the highest test concentrations (1400-1500 mg/L). Daphnids are a minor but sensitive part of the zooplankton community. Therefore, at TDS concentrations of 1700 mg/L, we might expect reduced abundance or possibly loss of sensitive zooplankton species, but little effect on more tolerant invertebrates or fish.

Since we do not and cannot have site-specific toxicity testing on all resident zooplankton species, we cannot be sure which species (if any) may be as sensitive to TDS as the two tested daphnids. Of the dominant zooplankton taxa (rotifers and copepods) the rotifer test data (one species) do not indicate such sensitivity. There are no site-specific toxicity test data for copepods. Galat and Robinson (1983) reported data for two species from a high

TDS lake (5,350 mg/L, mainly NaCl) and found effects at levels of 8,500 mg/L. Due to different ion ratios it is difficult to extrapolate from this result to Snap Lake.

The toxicity test data for phytoplankton species (*P. subcapitata* and *N. pelliculosa*) suggest growth stimulation rather than inhibition, at least at moderate TDS concentrations, with somewhat less stimulation at the highest TDS test concentrations (1,000-1,500 mg/L). While toxicity testing has not extended to the highest predicted TDS levels of 1700 mg/L, it is unlikely that these TDS levels would adversely affect the phytoplankton community.

Overall, there likely would be adverse effects on sensitive zooplankton taxa at the highest predicted TDS concentrations of 1700 mg/L in Snap Lake, and in downstream lakes 1 and 2 at predicted peaks around 1400 mg/L. The EAR predicted minor changes in the zooplankton community, but no loss of species. There is no evidence that TDS effects at the highest predicted exposure levels will extend beyond this threshold of significance, but major changes in the zooplankton community cannot be ruled out.

4.2 Chloride

DBCI (2013g) has predicted that chloride will exceed the SSWQO of 388 mg/L in the diffuser area as early as 2016 for the worst-case scenario (upper bound inflow, arithmetic mean TDS) and will reach this objective by 2026 for the best-case scenario (lower bound inflow, geometric mean TDS). Predicted peak concentrations will be about 800 mg/L for the worst-case scenario and 390 mg/L for the best-case scenario. Peak concentrations in the main basin will be only slightly lower.

As noted in Section 2.2, the SSWQO 388 mg/L is based on the IC25 vs hardness relationship for the daphnid *C. dubia*, adjusted down to represent an estimated 5^{th} percentile of the SSD, and evaluated at a hardness of 160 mg/L, beyond which there is no apparent benefit of hardness (Elphick et al., 2011a). At chloride levels of 388 mg/L, therefore, we might expect the most sensitive 5% of species to show a 25% impairment of growth or reproduction. At higher chloride levels we expect more species to be affected, and more impairment of the most sensitive species.

If we accept the slope of the hardness relationship from Elphick's data for *C. dubia*, and use that slope to adjust IC25 values for all species to a hardness of 160 mg/L, we find that two species have IC25 values below the predicted peak chloride concentration of 800 mg/L. These are both daphnids (*C. dubia* and *D. magna*). Thus it is likely that daphnids would be affected at the highest predicted chloride levels, while other species tested by Elphick may not be measurably affected. Of the dominant zooplankton taxa in Snap Lake (rotifers and copepods), from Elphick's data for the rotifer *B. calyciflorus,* we would not expect adverse effects at the predicted peak chloride level of 800 mg/L. Similarly, for the two copepod species studied by Galat and Robinson (1983), we would not expect adverse effects at this level.

Overall, there likely would be adverse effects on sensitive zooplankton taxa at the highest predicted chloride concentrations of 800 mg/L, and in downstream lakes 1 and 2 at peaks around 660 mg/L. The EAR predicted minor changes in the zooplankton community, but no loss of species. There is no evidence that chloride effects at the highest predicted exposure levels will extend beyond this threshold of significance, but major changes in the zooplankton community cannot be ruled out.

5.0 ASSESSMENT OF MITIGATION MEASURES

Are there feasible mitigation measures that can be implemented at the Snap Lake mine site that will either ensure that contaminants do not exceed WQOs, or will minimize effects to the aquatic receiving environment?

DBCI (2013g) has proposed a mitigation strategy for TDS and chloride, which includes segregation of dirty and clear water, and WTP expansion. In the Technical Session it was clarified that segregation of dirty and clear water would not be feasible. For nitrate, DBCI (2013c) has proposed to review blasting practices, explosives loading/storage practices, and dilution practices, and to consider feasibility of nitrate treatment. The mitigations have not been described or evaluated in sufficient detail to judge their effectiveness. Pilot studies are planned but their outcome is uncertain. Meaningful predictions of receiving water quality with implementation of the mitigations cannot be made until the mitigation plans are better defined.

Prior to the Technical Session, in March 2014, DBCI was asked to provide predictions of Snap Lake water quality with mitigations in place. Model results were provided in April, 2014, showing that if EQCs are met, then WQOs will not be exceeded in Snap Lake. Since we are confident that the model is reasonably accurate, and either realistic or conservative, we are also confident that if DBCI meets their proposed EQCs, the proposed WQOs will be met in Snap Lake.

This does not address the question of whether the conceptually proposed mitigations, when plans are finalized in detail, will be sufficient to enable the mine to meet the proposed EQCs. However, the Mine Water Treatment Plant Alternatives Evaluation (CH2MHill, 2012) suggests that TDS, chloride and nitrate removal efficiencies greater than 90% are possible using reverse osmosis technology. Thus, without consideration of economics, it should be feasible in theory to achieve EQCs in the treated mine effluent.

6.0 ASSESSMENT OF EQC CALCULATIONS

Is the proposed method of calculating EQC appropriate to meet the dual objective of minimizing waste discharge and protecting downstream water uses? Related to this: Is the dilution factor used by DBCI appropriate? Are the loss rates used for nitrite and ammonia reasonable and conservative? Overall, has DBCI used appropriate methodology to calculate their proposed EQCs?

The proposed method of calculating EQC is presented in DBCI (2013f). It is based on the concept of Waste Load Allocation (WLA), which DBCI has defined as an effluent quality concentration that will lead to a steady-state concentration at the WQO in Snap Lake. DBCI proposed to calculate EQCs for various measurement periods based on statistical considerations that would allow this WLA to be achieved. The periods under consideration are daily, monthly, and annually, leading to the following EQCs: Maximum Daily Limit (MDL), Average Monthly Limit (AML), and Annual Loading Limit (ALL). This method is outlined in separate guidance documents from Alberta Environmental Protection (1995) and the US EPA (1991), and is expected to allow WQOs to be met at or beyond the mixing zone boundary in Snap Lake at all times.

The first of the dual objectives outlined above, concerns whether or not this method will minimize waste discharge. We do not expect this method to minimize waste discharge; it is intended only to provide EQCs that allow WQOs to be met in the receiver. Loadings to Snap Lake will be allowed to increase for chloride, fluoride, ammonia, nitrite, and sulphate. DBCI has demonstrated that WQOs are predicted to be met in Snap Lake if the proposed EQCs are met in effluent.

The second of the dual objectives outlined above is the protection of downstream water uses. The method has been endorsed by regulatory agencies (Province of Alberta, US EPA) for protection of human health and ecological receptors, depending on the water quality objective used. As such, if the WQOs are protective, the proposed EQC calculation method is considered to meet the objective of protecting downstream water uses.

As part of our review, we undertook a check of DBCI's EQC calculations using Microsoft Excel. All parameters required for EQC calculations were provided by DBCI in the EQC report with the exception of the coefficients of variation of the COCs. We therefore back-calculated the coefficients of variation for the COCs using Excel's "goal seek" function and ensured that the MDLs and AMLs were calculated correctly. Based on this analysis, DBCI appears to have implemented the EQC calculation process according to the regulatory guidance with no mathematical errors. We also double-checked the tables of hardness-based calculations in Appendix I of the EQC report, using the sulphate calculation spreadsheet provided by DBCI, and verified that those calculations were also performed with no mathematical errors.

6.1 Dilution Factor

The definition of the dilution factor in the EQC report has led to considerable confusion on the part of the reviewer. DBCI has clarified since the release of the EQC report that the dilution factor of 12 originally appearing in the report was intended to be 11 instead, and that the definition of the dilution factor in the EQC report was not the same as the definition of the dilution factor in the 2012 Plume Characterization study despite the numerical similarity of the two quantities. We accept these clarifications.

In the EQC report, DBCI defines the dilution factor as the ratio of the number of volumes of lake water to each volume of effluent in the diffuser based on a mass balance of the mixing zone in Snap Lake only. This value was set at 11 based on hydrodynamic considerations using the CORMIX model. From a purely hydrodynamic perspective, then, 11 volumes of lake water mix with every volume of effluent in the diffuser, regardless of the concentrations of the COCs in lake water, and the factor of 11 thus remains unchanged over time even if COC concentrations in Snap Lake change.

DBCI uses this dilution factor consistently and correctly through the EQC derivation process.

6.2 Loss Rates for Ammonia and Nitrites

DBCI did not use explicit loss rates when calculating EQCs for ammonia and nitrites. However, the equation presented in the EQC report for these parameters (Equation 5), contains an implicit assumption that ammonia and nitrite concentrations will remain at baseline levels in Snap Lake over time. This implicit assumption directly contradicts the Snap Lake Hydrodynamic Model results, which predict a steady increase of ammonia in the Lake even with loss mechanisms in place (no predictions were available to verify this trend for nitrites). We recommend that DBCI re-derive the EQCs for ammonia and nitrites to account for accumulation of these chemicals in Snap Lake, and for losses consistent with the Snap Lake model.

6.3 Overall Assessment of EQC Method

Overall, the EQC calculation methodology from Alberta (1995) and US EPA (1991) was considered to be appropriate. EQCs for ammonia and nitrites should be re-derived to take loss terms into account consistent with the Snap Lake model.

7.0 References

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